Data-driven performance assessments for river channel restoration schemes

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Whilst registered as a candidate for the above degree, I have not been registered for any other research award. The results and conclusions embodied in this thesis are the work of the named candidate and have not been submitted for any other academic award.

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Abstract

River restoration is a developing global industry and science working to improve river health. Monitoring river restoration projects is critical to confirm that this practice is benefitting river health. Data-led monitoring has often been neglected due to resource constraints. Technological advancements have recently presented opportunities to improve the uptake of data-driven performance assessments for river restoration schemes. However, there appear to be few examples of these technologies being applied outside of academia. Therefore, this research aims to explore and present guidance on how cost-effective data collection, analysis and communication of geomorphological and physical habitat datasets may be routinely undertaken within industry.

A review of emerging technologies suggested that the Acoustic Doppler Current Profiler may be an effective tool for river restoration monitoring. The feasibility of this was evaluated by undertaking a data-driven performance assessment of the River Rother Habitat Enhancement Scheme, West Sussex. The scheme was assessed over an 18-month period and was found to be successful in achieving its overall objective of improving spawning habitat restoration within the targeted reach. Through utilising this technology and catchment baseline data, recommendations for the future sustainable management of the River Rother were outlined.

Data collection using the Acoustic Doppler Current Profiler was easy and efficient but the data processing and analysis components of this research required a significant investment of time and technical knowledge. This is likely to be a substantial barrier for widespread data-driven performance assessments beyond academia. The future development of open source software may go some way to alleviate these issues and improve the feasibility of such monitoring approaches.

High resolution datasets afford the opportunity for more accurate results and the development of excellent visual dissemination tools. These may foster learning amongst both technical and non-technical stakeholders. This thesis presents the concept of a performance tracking framework for river restoration schemes relative to their objectives. The concept is presented such that, with development, it could be integrated with existing learning platforms to improve opportunities for non-technical experts to track river restoration performance over time and highlight any needs for further restoration.

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Abbreviations

2D – Two dimensional

3D – Three dimensional

ADCP – Acoustic Doppler Current Profiler

ADV – Acoustic Doppler Velocimeter

ANOVA – Analysis of Variance

AOD – Above Ordnance Datum

ARRT – Arun and Rother Rivers Trust

ASCII - American Standard Code for Information Interchange

A-STAR – Accounting for Sediment Transfer along the River Rother

BA – Before and After Assessment

BACI – Before, After, Control and Impact Assessment

BBC – British Broadcasting Corporation

BGS – British Geological Society

MBACI – Multiple Before, After, Control and Impact Assessment

CAA – Civil Aviation Authority

CEH – Centre for Ecology and Hydrology

CRF – Catchment Restoration Fund

CV – Coefficient of Variation

CWA – Clean Water Act

DEM – Digital Elevation Model

Defra – Department for Agriculture and Rural Affairs

DGPS – Differential Global Positioning System

DOD – DEM of Difference

DSF – Decision Support Frameworks

DTM – Digital Terrain Model

EA – Environment Agency

ECM – Electromagnetic Current Meter

ELJ – Engineered Log Jams

EU – European Union

GPS – Global Positioning System

GUI – Graphical User Interface

JBA – Jeremy Benn Associates

HMID – Hydromorphological Index of Diversity

HSI – Habitat Suitability Index

HCA – Hydraulic Clustering Analysis

HLS – Higher Level Stewardship

ICUN – International Union for the Conservation of Nature

MRRT – Manual of River Restoration Techniques

MWA – Moving window analysis

NERC – National Environmental Research Council

NRA – National Rivers Authority

NRRI – National River Restoration Inventory

NRRSS – National River Restoration Science Synthesis

ONEMA – French National Agency for Water and Aquatic Environments

PBA – Peter Brett Associates

PHABSIM – Physical Habitat Simulation System

PHH – Physical Habitat Heterogeneity

PHP – Physical Habitat Performance

PPA – Post-Project Appraisal

PRAGMO – Practical Guidance for River Restoration Monitoring

QCD – Quantile Coefficient of Dispersion

RCHARC – Riverine Community Habitat Assessment and Restoration Concept

REF – Research Excellence Framework

RHES – River Rother Habitat Enhancement Scheme

RHS – River Habitat Survey

RRC – River Restoration Centre

RMSE – Root Mean Squared Error

RPA – Rural Payments Agency

RTK – Real Time Kinematic

SBAS – Satellite Based Augmentation System

SMART – Specific, Measurable, Achievable, Realistic and Time-bound (objectives)

SMART – Sediment Mitigation Options for the River Rother

SNDP – South Downs National Park

SNDPA – South Downs National Park Authority

SFM – Structure-from-Motion

SW – Southern Water

TIN – Triangle Irregular Network

TSL – Terrestrial Laser Scanner

UAV – Unmanned Aerial Vehicle

UoC – University of Chichester

UoN – University of Northampton

UoO – University of Oxford

UoP – University of Portsmouth

UK – United Kingdom

US – United States

USGS – United States Geological Survey

VMT – Velocity Mapping Toolbox

WFD – Water Framework Directive

WUA – Weighted Usable Area

Dissemination

Formal Presentations

Cox. J. (2013). Eco-hydraulic patch dynamics for river restoration schemes. Poster presentation at the Science Faculty Post-Graduate Research Conference, University of Portsmouth. Portsmouth.

Cox. J. (2013). Eco-hydraulic patch dynamics for river restoration schemes. Oral presentation in the Department of Geography's Seminar Series, University of Portsmouth, Portsmouth.

Cox. J. and Soar, P.J. (2013). Performance of an artificial riffle in the River Rother, West Sussex, and implications for habitat. Poster presentation at the British Society for Geomorphology Conference, University of Manchester, Manchester.

Cox. J. and Soar, P.J. (2014). Resilience and morphological performance of an artificial riffle on the river Rother, West Sussex. Oral presentation at the Healthy Environment, Healthy Lives: Towards a 21st Century Park - South Downs Research Conference, South Downs Centre, Midhurst.

Cox, J. and Soar, P.J. (2015). Application of an Acoustic Doppler Current Profiler for river restoration monitoring and appraisal. Oral Presentation at the River Restoration Northwest Symposium, Skamania Lodge, Washington, US.

Cox, J. and Soar, P.J. (2015). Experiences of restoring physical habitat with an artificial riffle. Oral presentation at the River Restoration Centre Annual Networking Conference, Whiittlebury Hall, Northampton.

Cox. J. and Soar, P.J. (2015). 'Lessons for the future: Learning from river restoration in the National Park'. Oral presentation at the 'Embracing the future: Managing the environment, heritage and change in the South Downs National Park - South Downs Research Conference, South Downs Centre, Midhurst.

Cox. J. (2017). Opportunities in Geomorphology. Oral presentation at the Water & Environment Practice of Ricardo Energy and Environment Consultancy Awayday, Chester Zoo, Chester.

Reports

Cox, J. and Soar, P.J. (2016) Engaging with river restoration: the performance of habitat enhancement in the River Rother, West Sussex, as a rich learning resource. Report to the South Downs National Park Authority. University of Portsmouth.

Cox, J. and Soar, P.J. (2017) Accounting for sediment transfer along the River Rother, West Sussex. Report to Southern Water, the South Downs National Park Authority, the Environment Agency and the Arun and Rother Rivers Trust. University of Portsmouth.

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for Eileen Cox

We are such stuff as dreams are made on;

and our little life is rounded with a sleep.

-Shakespeare, Tempest, Act 3

1 Introduction

1.1 Introduction

This chapter introduces the key concepts explored within this thesis which is undertaken in the context of river restoration monitoring practices. The overarching drivers and development of the broad river restoration discipline are presented here. River restoration has evolved in such a way that it may be used to improve the environmental, social and economic functioning of rivers. However, this thesis is primarily concerned with the restoration of physical habitats; an environmental feature of rivers. River restoration is a relatively young practice and an even younger science, consequently learning and adaptive management are critical to ensure restoration practices are contributing to an improvement in the condition of our rivers. This chapter highlights the importance of monitoring and explores examples of where monitoring has demonstrated both highly effective and less effective practices.

The current state of river restoration monitoring is explored here. Learning may be facilitated by data-driven performance assessments such as those collecting high-resolution physical habitat data. Resource constraints typically appear to be a significant barrier to river restoration monitoring, such that monitoring programmes (particularly over the longer-term) are rarely completed. Emerging technologies may have the potential to improve the likelihood of data-driven performance assessments, but there appears to be limited guidance and feasibility studies of their use in practice. This chapter highlights that effective river restoration monitoring for learning and adaptive management is dependent on data collection, analysis and dissemination. Consequently, providing an assessment of the feasibility of novel technologies for river restoration monitoring practice in relation to these three core components is an overall aim of this thesis. The later sections of this chapter outline the research objectives and the overall structure of this thesis. The research documented within this thesis is very applied and has the potential to influence the river restoration industry. Accordingly, the research is centred around a case-study river restoration scheme which is explored in detail within this thesis and utilised to make recommendations for improving river restoration practices.

1.2 River Restoration

The Anthropocene is a geological epoch defined by the overwhelming influence of human activity on the formation of earth's surface (Zalasiewicz et al., 2013). The case for this new epoch in geological history is particularly relevant in the study of fluvial geomorphology (Brown et al., 2016), since rivers are an economically important natural resource that globally provide a range of goods and services which support human society (Fig. 1.1) (Palmer et al., 2005; Harvey & Clifford, 2009; Posthumus et al., 2010; Gilvear et al., 2013; Ormerod, 2014). These goods and services, otherwise known as ecosystem services (Gilvear et al., 2013), have been used for thousands of years with channel modification for economic gain and protection dated back to Ancient Egypt (Said, 1983; Downs & Gregory, 2004). Extensive historical use of ecosystem services in the United Kingdom (UK) is evidenced by recent archaeological investigations on lowland floodplains. Boats and fish weirs have been found amongst artefacts of Bronze Age settlements in palaeochannels of the River Nene, Cambridgeshire (Malim et al., 2015) and the River Trent, Derbyshire (Howard et al., 2017). These archaeological findings indicate that worldwide river systems and human society have had a long and complexly intertwined history.

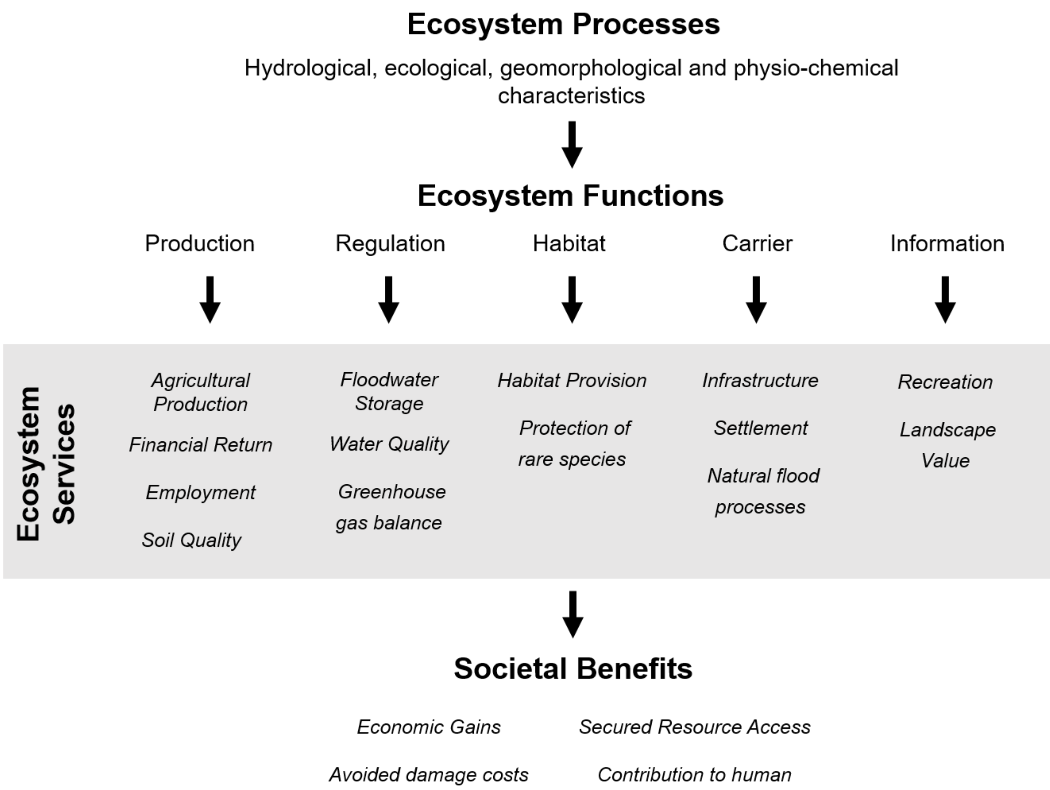


Figure 1.1 Ecosystem Services (adapted from Posthumus et al., 2010)

In more recent history, the industrial revolution and particularly the post-World War II drive for self-sufficiency has seen substantial channel modification for flood defences to protect agricultural land and urban areas in the UK (Posthumus et al., 2010). In the United States (US) over the last 300 years, more than 75,000 dams have been constructed to provide multiple benefits to society including flood defence and navigation (Graf, 2005; 2006). Changes within river catchments such as those described above are examples of disturbance events in an environmental system, which may prompt system-wide changes in geomorphological and ecological processes.

The system response to disturbance events has been simply explained through equilibrium theory (Hack, 1960), however, it is acknowledged that system response is typically more complex, stochastic and non-linear (Phillips, 1992; 2006; 2015). It has been argued that over long time scales (hundreds to thousands of years), the fluvial system has been in a state of dynamic equilibrium whereby the system follows a trajectory towards a stable condition (Schumm, 1975). However, a disturbance event may cause a system to cross a threshold onto a new trajectory towards a different stable condition (Fig. 1.2) which is known as dynamic metastable equilibrium (Schumm, 1975). For example, sediment mining and dam construction reduced the sediment supply on the Brenta River, Italy, which contributed to a reduction in the braiding intensity and a wandering planform (Surian and Rinaldi, 2003). Natural disturbance events which cause change on a par with human-induced changes to system behaviour within the Anthropocene are typically cataclysmic. For example, the 1980 eruption of Mt. St. Helens transformed the once mildly sinuous gravel-bed North Fork Toutle River into a sand-dominated braided river (Simon and Thorne, 1996; Zheng et al., 2014).

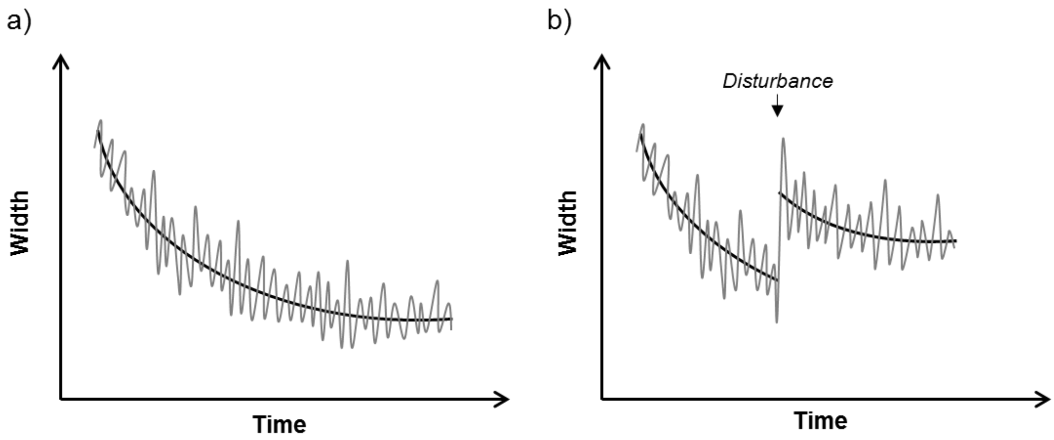


Figure 1.2 Trajectories in environmental systems a) dynamic equilibrium, b) dynamic metastable equilibrium in response to a disturbance event (adapted example of channel width from Schumm, 1975)

Disturbances to the fluvial system throughout the Anthropocene have interfered with geomorphological and hydrological processes that have previously supported a diverse ecological community. For instance, a decline in macroinvertebrate species diversity has been associated with decreased flows and sediment supply from dam construction (Vinson, 2001). Conversely, increases in fine sediment as a result of agriculturally induced soil erosion also has the potential to be deleterious on the lotic community (Wood and Armitage, 1997; Kemp et al., 2011; Jones et al., 2012). Ecological communities are in decline through the degradation but also fragmentation of habitats (Woolsey et al., 2007; Smith et al., 2014) and it has been claimed that riverine ecosystems are experiencing extinction rates 5 times that of terrestrial ecosystems (Ricciardi and Rasmussen, 1999; Bernhardt et al., 2005).

Some strategies to mitigate against these potential deleterious disturbance events have been successful (Vinison, 2001). As a result, the river restoration industry has gained momentum over the last few decades largely fuelled by the demands of policies across the world aimed at improving river condition such as the European Union (EU) Water Framework Directive (WFD, 2000/60EC) and United States Clean Water Act (CWA, 1979) (Lave et al., 2010). However, national agendas are also highly influential in river restoration practice. For example, in the UK, environmental policies such as ‘Making Space for Water’ (Defra, 2005) and agri-environmental schemes such as Higher Level Stewardship (HLS) (Defra, 2004) have been influential in river management practice alongside the WFD (England et al., 2008).

The practice of river restoration has rapidly developed since the mid-20th century (Ormerod, 2014) and has become a substantial global billion-dollar industry (Bernhardt et al., 2005). However, the overarching definition of river restoration has been heavily contested; Cairns (1990) defined river restoration as “the complete structural and functional return to a pre-disturbance state”. In the debate of the Anthropocene’s existence, a case has been made that within the fluvial system, human impacts are likely irreversible (Brierley & Fryiars, 2005; Brown et al., 2013). Therefore, a more appropriate aim for river restoration is to maintain a sustainable environment in the context of prevailing catchment conditions (Hilman & Brierley, 2005; Wohl, 2005). This may be achieved by a suite of conservation techniques as described by Brookes (1996) such as “restoration, rehabilitation, enhancement, and/or creation”. In this thesis, the holistic definition of river restoration by Wohl et al., (2005) is favoured, namely, “assisting the establishment of improved hydrologic, geomorphic and ecological processes in a degraded watershed system and replacing lost, damaged or compromised elements of the natural system”.

The nature of river restoration has evolved since pioneering projects, from an engineering approach (i.e. deterministic and reach scale restoration objectives) to an ecosystem approach (i.e. probabilistic, catchment scale and interdisciplinary objectives) (Hillman & Brierley, 2005; Kondolf et al., 2007). Restoration techniques have steered away from form-based or single-species led objectives towards more process-based and wider ecosystem led objectives (Palmer, 2008). Reservations have been expressed about catchment scale restoration as the outcomes of restoration at the reach scale are not yet fully understood (Jansoon et al., 2004; Ormerod, 2004). Presently, catchment scale restoration is mostly limited to demonstration projects with the majority of restoration still focused at the reach scale (Skinner & Bruce-Burgess, 2005; Smith et al., 2014; Castillo et al., 2016), as factors such as landowner permission often mean restoration is still largely opportunistic (Palmer, 2008). However, the consideration of catchment processes is fundamental to successful restoration (Bernhardt et al., 2011).

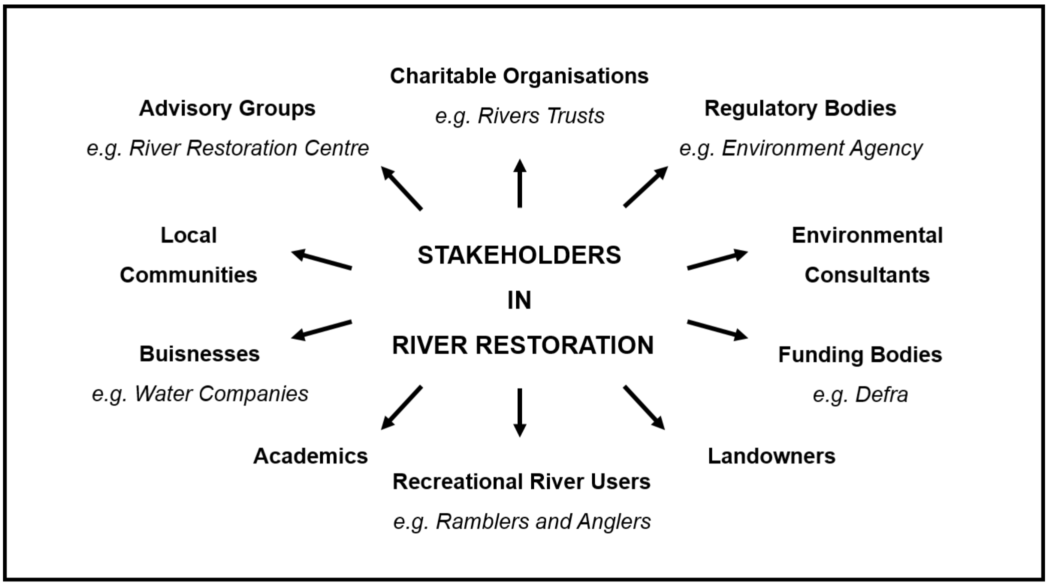


Figure 1.3 Stakeholders in River Restoration with examples from UK river restoration

River restoration has a widening range of stakeholders (Fig.1.3) as a result of catchment scale restoration and interdisciplinary objectives (Smith et al., 2014) and, as such, is increasingly becoming a social science (Lave, 2015). Stakeholder participation in the decision-making process is a requirement of policies such as EU WFD and US CWA, and as a result, participation has emerged as a key agenda in many river restoration projects (Carr et al., 2012; Smith et al., 2014). Participatory approaches are particularly important in regions where catchments span multiple jurisdictional boundaries (Akamani and Wilson, 2011) and/or land of indigenous populations (Bark et al., 2012; Jackson et al., 2012).

There is typically wide public support for conservation activities, as rivers can be particularly emotive as they hold social and cultural value. However, there can also be local opposition to river restoration particularly when the removal of structures are proposed (Fox et al., 2016; Keilty et al., 2016). The importance of stakeholder participation has been reflected by the International Union for the Conservation of Nature UK (IUCN) who commented that “the reward of river restoration is naturally functioning rivers that support improved biodiversity while bringing benefits for a society that is re-engaged with rivers” (Addy et al., 2016).

1.3 Restoration of Physical Habitat

Reflecting the increasingly multidisciplinary nature of river restoration, the range of techniques used for restoration have increased over time. In particular, in-stream habitat improvement techniques have increased in popularity over the last decade (Castillo et al., 2016). The improvement of physical habitats through modification of geomorphological processes is a common design objective of restoration (Downs and Thorne, 2000). Unless otherwise stated, the term ‘river restoration’ henceforth refers to the practice of intervention for improving the condition of physical habitat. Physical habitats are defined as the abiotic environment used by an organism, excluding physiochemical properties such as temperature. These spaces are controlled by the complex interaction between geomorphology and hydrology (Fig. 1.4) of river system (Maddock et al., 1999).

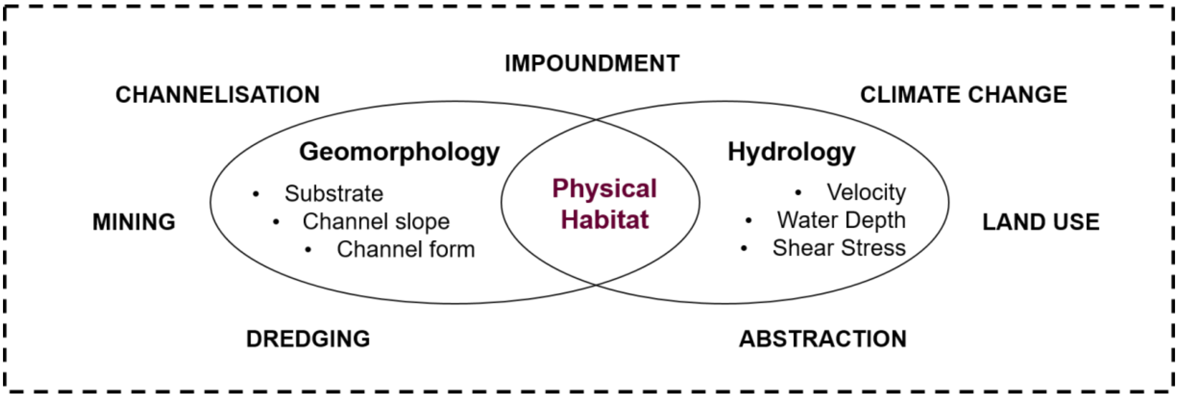


Figure 1.4 Determining factors on physical habitats, adapted from Maddock (1999) to include anthropogenic factors for consideration

The geomorphology of a river system constitutes the forms and materials of the physical habitat including characteristics such as channel form, slope and substrate (Maddock et al., 1999). The hydrology of a river system controls discharge which influences the velocity, water depth and shear stress within the river channel (Maddock et al., 1999). Such physical habitat features are dynamic over time and space as a function of discharge (Wallis et al., 2012). The mosaic of physical habitats within the river system is known as the riverscape; a diverse array of physical habitats within the riverscape is required to provide a range of ecological niches for different species and their respective life stages (Ward, 1998). The processes in the fluvial system which support a diverse riverscape are intrinsically connected in four dimensions; longitudinal (downstream), lateral (floodplain), vertical (hyporheic zone) and through time (Petts & Amoros, 1996).

Anthropogenic activities have typically degraded the riverscape, and thus provision of physical habitats, through altering the geomorphology and hydrology of the Earth's surface (Ward, 1998). Activities such as dredging, mining and channelisation have predominantly resulted in a direct alteration of geomorphology, whereas changes in hydrology are more frequently associated with events such as climate change and shifting land use patterns (Fig. 1.4). However, the geomorphology and hydrology of the fluvial system are innately linked and therefore, the modification of either geomorphological or hydrological processes will likely result in a degree of change in the other.

The physical habitat approach to river restoration assumes habitat heterogeneity improves ecological status through the provision of a variety of ecological niches (Kemp et al, 1999). However, this 'build and they will come' approach is a subject of contention in the literature. Physical habitat studies have reported mixed results on ecological performance following improvement on habitat heterogeneity. Studies have noted ecological improvements following improvements in habitat heterogeneity (e.g. Lorenz et al., 2013, Thomas et al., 2015), other studies have noted a limited ecological response (e.g. Harrison et al., 2004, Haase et al., 2013) and many studies have proven inconclusive citing insufficient data (Brooks et al., 2002; Miller et al., 2010; Palmer et al., 2010; Perderson et al., 2014). A range of other limiting factors such as water quality, habitat fragmentation, and temporal persistence of physical habitat can also influence ecosystem recovery (Pretty et al., 2003; Gostner et al., 2013). These factors are particularly important if they are operating beyond the reach scale within the catchment (Stoll et al., 2016). Therefore, there is a need to understand physical habitats and ecosystem functions better to inform river restoration (Palmer and Bernhardt, 2006; Elosegi & Sebater, 2013). River restoration is a young practice and younger science, and the effects of intervention may not be realised in the short-term. Consequently, long-term monitoring and continual learning in both science and practice are needed (Roni et al. 2008).

1.4 Learning from River Restoration

The perception of river restoration as a force of ‘good’ for the riverine ecosystem often remains unchallenged in the absence of rigorous monitoring of project performance. This may lead to unsustainable practices or the potential for continued degradation of aquatic ecosystems (Kondolf & Micheli, 1995; Downs & Kondolf, 2002). This is particularly relevant if restoration has improved aesthetics but failed to restore function (Wohl, 2005). As a society, we have an inherent preference for aesthetically pleasing single meandering channels (Kondolf, 2006), and it has been observed that significantly more single thread meandering channels exist in their natural state than braided or anastomosed rivers (Tockner and Standford, 2002).

As an example of this, in a study of rivers as play spaces for children, Tunstall et al., (2004) noted that some children suggested the removal of riparian vegetation (long grass) would improve the quality of the river as a play space. This ‘image’ was likely modelled on urban landscapes and parkland in which vegetation is routinely managed. In addition, the increasing spatial restriction on play from home limits the potential exposure to and exploration of river environments by children (Tapsell, 1997). These findings highlight how the perception of rivers can be influenced from a young age and a lack of interaction with the natural environment can exacerbate pre-conceived ideas of river aesthetics. Modern urban landscapes are not the only environment which can influence the perception of a natural river system. Iconic British nature-esque landscapes in many stately homes across the UK by Lancelot “Capability” Brown (1716-1783) regularly feature single-thread meandering water features based on Hogarths “Line of Beauty” (Kondolf, 2006, Kondolf et al., 2016). However, Poldark *et al.* (2013) noted the persistence of these artificial water features over the last two centuries has been as a result of regular maintenance, as the designs had not accounted for sediment transport processes.

A prime example of restoration restoring form without process is illustrated by Soar and Thorne (2001) in the restoration of Whitemarsh Run, Maryland, US. Previously channelised to enable floodplain development and flood alleviation, the low sinuosity channel of Whitemarsh run was restored in 1996 to a more sinuous planform. The design, however, has since proved to be unsustainable. Widespread aggradation within the restored area occurred due to the failure to account for exacerbated sediment inputs from agricultural land use and the local reduction in slope from the increased channel sinuosity in the restoration design. Similarly, other schemes which have applied the highly contested Rosgen (1994) classification to identify stable hydraulic geometry of river channels have failed because they have not appropriately considered local sediment transport processes, for example the restoration of Deep Run, Maryland, US (Downs & Kondolf, 2002; Smith and Prestegaard, 2005; Kondolf, 2006; Simon et al., 2007; Lave, 2009).

These examples illustrate that river restoration is not an exact science. Academic participation in river restoration has gained momentum over the last few decades to improve the science basis of this practice (Smith et al., 2013; Wohl et al., 2015). In this time, scientists have contributed technical guidance to assist river restoration delivery (e.g. Thorne et al., 1997; Soar & Thorne, 2001; Sear et al., 2010; Simon et al., 2013). Ultimately, the fluvial system is still largely indeterminate with design equations based on empirical relationships such as hydraulic geometry equations (Soar & Thorne, 2001). Consequently, uncertainty in restoring physical habitat is to be expected. However, through experiential learning in an adaptive management process, the level of uncertainty can be reduced as we gather knowledge on the strengths and weaknesses of projects (Haney and Power, 1996; Levine, 2004; Hillman and Brierley, 2005; Palmer et al., 2007).

River restoration protocols such as Thorne (2002) and Skinner and Bruce-Burgess (2005), are examples of adaptive management in river restoration (Fig. 1.5). Key components in these protocols which facilitate learning are the objective setting, monitoring and dissemination processes. The dominant need for rigorous monitoring of river restoration projects is to collect information to inform the design of future projects (Clarke et al., 2003, Palmer et al., 2005; Woolsey et al., 2007). In addition to learning potential, the ability to demonstrate the ecological success of restoration schemes is needed and may improve the likelihood of further funding for river restoration (Woolsey et al., 2007; Mainstone and Wheeldon, 2016). Furthermore, in the EU, monitoring is a policy requirement of the WFD to demonstrate waterbody quality does not decline further (England et al., 2008; Matthews et al., 2010).

Monitoring has the potential to identify not only ineffective restoration but also successful measures which could be adopted in future designs. Monitoring of restoration on the River Idle in 1996 identified deflectors as an effective approach for reintroducing morphological and habitat diversity to a channelised reach (Downs & Thorne, 1998; 2000). Similarly, monitoring of the Kissimmee River Restoration Project, Florida, revealed that the re-establishment of a meandering planform in response to channelisation in the 1960s had been successful at improving sediment transport and point bar development (Anderson, 2014). In the case of the Kissimmee Restoration Project, restoration has been undertaken in several phases and ongoing monitoring has informed later phases of restorative work in an adaptive management process (Koebel & Bosquin, 2014).

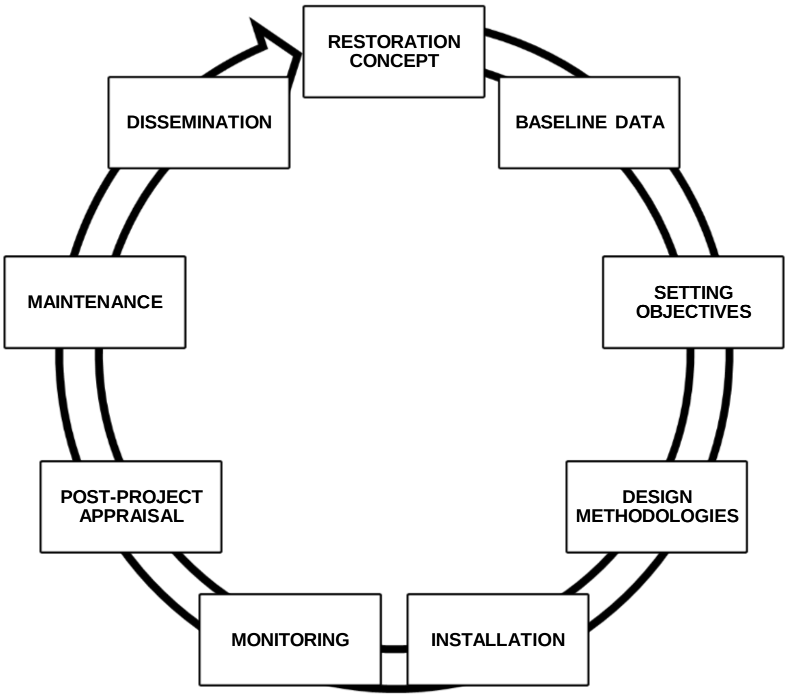


Figure 1.5 The River Restoration Protocol, adapted from Skinner and Bruce-Burgess (2005) to reflect river restoration as in the cyclical adaptive management process.

These examples highlight how monitoring may yield useful learning outcomes, but monitoring every project may be unfeasible. A targeted approach to monitoring, based on the scale and risk of the project may be more appropriate (England et al., 2008). Bernhardt et al. (2005) observed larger, higher risk projects in the US were more likely to be monitored, however, smaller scale restoration activities have accounted for a larger proportion of total river restoration expenditure. This may reflect the nature of river restoration as an opportunistic rather than strategic management strategy (Smith et al., 2013). It also highlights that the monitoring of smaller schemes is perhaps as important if not more as large scale projects for evaluating economic benefit, environmental impact and learning potential. Unfortunately, however, an overarching observation from river restoration literature and advisory groups is the lack of river restoration monitoring and dissemination (Wohl et al., 2015).

1.5 State of Monitoring

Many advisory bodies and working groups such as the US National River Restoration Science Synthesis (NRRSS, Bernhardt et al., 2007), the French National Agency for Water and Aquatic Environments (ONEMA, Morandi et al., 2015) and the UK River Restoration Centre (Smith et al., 2013) hold databases containing details of river restoration projects within their geographical remit. Reviews of databases such as the NRRSS (Bernhardt et al., 2005) and NRRI (Smith et al. 2013) have provided some valuable insights into river restoration practices such as prevalence of monitoring.

In a review of the US National River Restoration Science Synthesis (NRRSS) of 37 000 river restoration projects, only 10% of the projects undertook any form of monitoring (Bernhardt et al., 2005). Of a sub-set of these projects undertaken in California, it was found that 20% of the projects undertook monitoring, but only half of these stated the form of monitoring undertaken (Kondolf *et al*., 2007). The reasoning for monitoring has been clearly communicated earlier in this chapter (Section 1.3). However, there appear to be barriers preventing both academics and practitioners from undertaking effectual monitoring programmes.

In terms of engagement with monitoring by academics, research assessment exercises such as the most recent Research Excellence Framework (REF) in 2015 in the UK have changed the nature of academic reporting with a preference towards original research articles in peer-reviewed journals (Woodward, 2015). In light of these changes, monitoring and appraisals as 'applied research' may present a poor investment opportunity to academics (Dickens and Suding, 2013; Wohl et al., 2015). This may partially explain the lack of published monitoring results in peer-reviewed journals (Bernhardt et al., 2007; Garvey et al., 2010). However, in both the UK and US, funding bodies (such as National Environmental Research Council, NERC UK) have included 'impact' as an obligatory component of research projects (Gregory, 2014).

In the UK, NERC defines impact as "An effect on, change or benefit to the economy, society, culture, public policy or services, health, the environment or quality of life, beyond academia". Thus in future, the need to demonstrate impact in academic research may improve the appeal of applied research, such as river restoration monitoring and appraisal studies. Therefore, river restoration may see an increase in academic involvement as a result. However, in order to maximise the benefit from academic-practitioner collaborations, academics must integrate themselves as a stakeholder within river restoration projects (Poff et al., 2003). Also, to avoid the 'square peg-round hole' conundrum, the important questions that need answering must be clearly defined, else river restoration may see the application of poorly fitted research (Palmer, 2008).

In contrast to the academic study of river restoration, a lack of practitioner monitoring has been associated with three main factors. Firstly financial, labour and resource constraints can restrict the ability to monitor (Bash & Ryan, 2002, Kondolf et al., 2007; Dickens and Suding, 2013). The preference of funders for project delivery rather than monitoring has been identified as a key barrier to monitoring (Downs et al., 2011). Furthermore, funding is generally short-term (less than 5 years) and does not provide scope for long-term monitoring projects (Levine, 2004). Over the last decade in particular, the global economic crisis has seen cuts in government spending which has impacted on the river restoration practitioner (Raven et al., 2011). The IUCN review of river restoration practice in the UK and Republic of Ireland has called for longer-term (more than 5 years) government funding to support the river restoration process, including monitoring (Addy et al., 2016). The current lack of long-term funding has fostered the view within practice that ‘partnerships with local universities are likely to be the most cost-effective means of securing a lasting association between river restoration and monitoring’ (Mainstone and Wheeldon, 2016).

Secondly, due to a culture of fear, practitioners may be reluctant to participate in monitoring as it opens up work to a critical review (Dickens and Suding, 2013). This has been seen as a barrier in collaborative projects between industry and academia (Wilcock, 1997). Critical review, particularly negative reviews, could be perceived as a threat to future sources of funding. Finally, low participation in monitoring is associated with a lack of clear guidance on appropriate methods as well as a lack of tools to interpret meaningful results (England et al., 2008). For example, the geomorphological Post-Project Appraisal (PPA) defined by Downs and Kondolf (2002) utilises monitoring data to assess compliance with design as well as the performance of the scheme in meeting project objectives in relation to a pre-project or baseline condition. The guidance for undertaking a PPA is given in Thorne *et al.* (2010), however, there is little practical guidance or frameworks which enable the practitioner to evaluate monitoring datasets. Practical guidance for river restoration monitoring can be found in the grey literature (e.g. RRC, 2011). This provides guidance on data collection but provides limited guidance on data analysis to draw conclusions.

The continued prompts for a greater number of high-quality monitoring programs may correlate with a recent increase in the number of monitoring programmes in the US (Buchanan et al., 2012). However, many of these monitoring programs have been found to lack rigour (Vaughan et al., 2009; Buchanan et al., 2012; Gilvear, 2013). The state of monitoring in the UK has not been widely explored but the monitoring programme of the Shopham Loop restoration project (2004) on the River Rother (West Sussex, UK) was critically evaluated by the Environment Agency (EA). The monitoring programme yielded various ecological and geomorphological datasets yet the subsequent analyses of these datasets were found inconclusive.

The monitoring programme employed a cross-sectional approach to capture geomorphic change, but infrequent and inconsistent sampling resolution between surveys limited the ability to detect subtle in-channel geomorphic changes (Environment Agency, 2015). Physical biotope mapping was also performed post-restoration, yet no baseline or subsequent survey was completed to allow comparison (Environment Agency, 2015). Furthermore, channel recovery in response to restoration was only becoming apparent in 2009 so continued monitoring was advised, yet to the author's knowledge no further monitoring has been undertaken at the site to date. This example illustrates the importance of monitoring protocol design and that learning in river restoration can be impeded by both a lack of quantity and quality of monitoring surveys.

Poor objective setting has also been identified as a limitation of river restoration monitoring, as success cannot be measured without knowing the end goal (England et al., 2008). In a review of the US NRRSS database, only 20% of these projects had stated project objectives (Bernhardt et al., 2005). Poor objective setting at the project outset can lead to the collection of inappropriate data and thus the inability to appropriately assess the project. Therefore, there is a need to promote the existing guidance on 'what does success look like?' to facilitate objective setting and subsequent monitoring programme design. Practitioners guidance advocates the use of SMART (specific, measurable, achievable, realistic and time-bound) objectives (Doran, 1891; Woodward and Hollar, 2011; RRC, 2011). With an emphasis on the "measurable" in SMART objectives, the monitoring techniques used must be effective in evaluating the success of the restoration project (Brierley et al., 2010). There appears to be a dominance in the academic and grey literature to focus on biological monitoring studies over geomorphic assessments (Brierley et al., 2010). Whilst improved ecological status may be the end goal of many restoration schemes, many North American and European restoration projects focus on the delivery of improved habitat heterogeneity as restoration objective (Palmer et al., 2010). Therefore, in these situations, biological monitoring alone may not be effective in providing an effective evaluation of restoration objectives.

A lack of baseline survey data is also a common issue in the appraisal of river restoration schemes, as relative performance cannot be measured (Brierley et al., 2010). In Washington State, US, only half of the surveyed projects collected a baseline dataset (Bash & Ryan, 2002). Projects with an ecological focus were found more likely to have a baseline dataset than those with an engineering focus (Bash & Ryan, 2002). Consequently, baseline surveys in Washington State were found to be more highly comprised of ecological data than geomorphological data. Fish counts and habitat surveys accounted for 29% and 23% of baseline data respectively, whereas cross sections and substrate surveys accounted for only 13% and 6% of baseline data respectively.

This trend is mirrored in surveys of river restoration practices in Scotland (Gilvear et al., 2013) and France (Morandi et al., 2014). Ecological surveys of fish and invertebrates were not only found to be more prevalent but also more likely to span a longer pre-project period than physical or chemical monitoring surveys. Baseline monitoring for all disciplines most frequently occurred within 1-2 years prior to project implementation, and therefore the importance of pre-project long-term monitoring must also be highlighted in monitoring guidance (Morandi et al., 2014). When interviewed, US practitioners were found to be aware of the importance of the baseline survey and wider monitoring (Kondolf et al., 2007), however, funding, and not guidance was found to be a key constraint. Ideally, to improve the uptake in recording the pre-restoration condition, monitoring should be a core component costed into funding proposals by river restoration practitioners (Downs & Thorne, 1998). This should be done alongside improved communication with stakeholders that fund projects to raise awareness of the importance of monitoring.

Long-term monitoring programmes are rarer as funding may only be provisioned for a short period of time. For example, in the US funding for monitoring over a 2-year period may be typical (Wendt, 2015). Long-term monitoring programmes may also be more difficult to fund if the monitoring techniques are more resource intensive (Mainstone and Wheeldon, 2016). Monitoring via citizen science has become a more common component of river restoration schemes, as many schemes have a limited budget and using volunteers can be a cost-effective approach. The citizen science monitoring approach must be simplistic and not assume specialist knowledge. This has typically led to the generation of qualitative data to feed into the adaptive management process (Smith et al., 2014). Qualitative walkover surveys assessing the structural integrity of the scheme over time do not equate to information regarding ecological performance (Palmer et al., 2007). Citizen science monitoring schemes present an opportunity for evaluating low-risk projects, namely those using established restoration techniques in familiar environments. Frameworks to facilitate the delivery of citizen science river restoration monitoring are emerging and have shown some promising results, for example MoRPh (Shuker et al., 2017).

Higher risk projects, namely those using innovative techniques or established techniques which are applied to a new environment, may require a more critical monitoring programme to maximise the opportunities for learning (England et al., 2008). These higher risk projects often require intensive data driven monitoring leading into a rigorous evaluation process (England et al., 2008), but this will likely demand specialist knowledge beyond that of a volunteer. Higher resolution data collection of physical habitats is uncommon in river restoration as it is associated with resource intensive methods of data collection (Maddock, 1999). Recent technological advancements such as Structure from Motion (Woodget et al., 2014, 2017; Dietrich et al., 2016) and Acoustic Velocimetry (Sheilds et al., 2003) have improved the cost-effectiveness of collecting high-resolution data of physical habitat variables. However, these have seen limited application outside of academia in river restoration monitoring as these techniques are relatively novel and costly. The current monitoring guidance, such as Practical Guidance for River Restoration Monitoring (PRAGMO, RRC, 2011) and the Applied Guidebook of Geomorphology (Thorne et al., 2010), do not yet incorporate guidance on performing monitoring using these new technologies or procedures for evaluating their data. There also does not appear to be an equivalent framework for data-driven monitoring like MoRPh which was developed for citizen science monitoring based on academic research (Shuker et al., 2017).

Additionally, the communication of river restoration performance following a monitoring programme does not appear to have a standardised practice. The dissemination of findings to the wider river restoration community and beyond is crucial in the adaptive management processes. In contrast to other studies, Matthews et al. (2010) suggest that monitoring is routinely undertaken to assess project success, but a lack of standardised procedures for monitoring, evaluation and dissemination limits the ability to both communicate outcomes and compare results from different schemes (Castillo et al., 2016). Without appropriate methods to disseminate and ability of stakeholders to access learning outcomes from the analysis of monitoring datasets, these datasets are at risk of becoming 'dark data'. 'Dark data' is the output of scientific investigation which becomes 'invisible' to the wider community and is therefore underutilised (Heidorn, 2008).

Consequently, the integral components of a successful data-driven river restoration monitoring programme are summarised as efficient data collection, data analysis and data dissemination. These are explored further in Chapter 2. If stakeholders are provided with adequate resources or can form innovative partnerships to share resources, data collection, analysis and dissemination can foster learning amongst stakeholders and promote adaptive management. This is conceptualised in Figure 1.6. These three integral components should be undertaken by a range of stakeholders that hold the appropriate technical expertise. However, the application of data-driven river restoration performance assessments beyond academic study appears to be rare, and this is the focus of the research reported in this thesis.

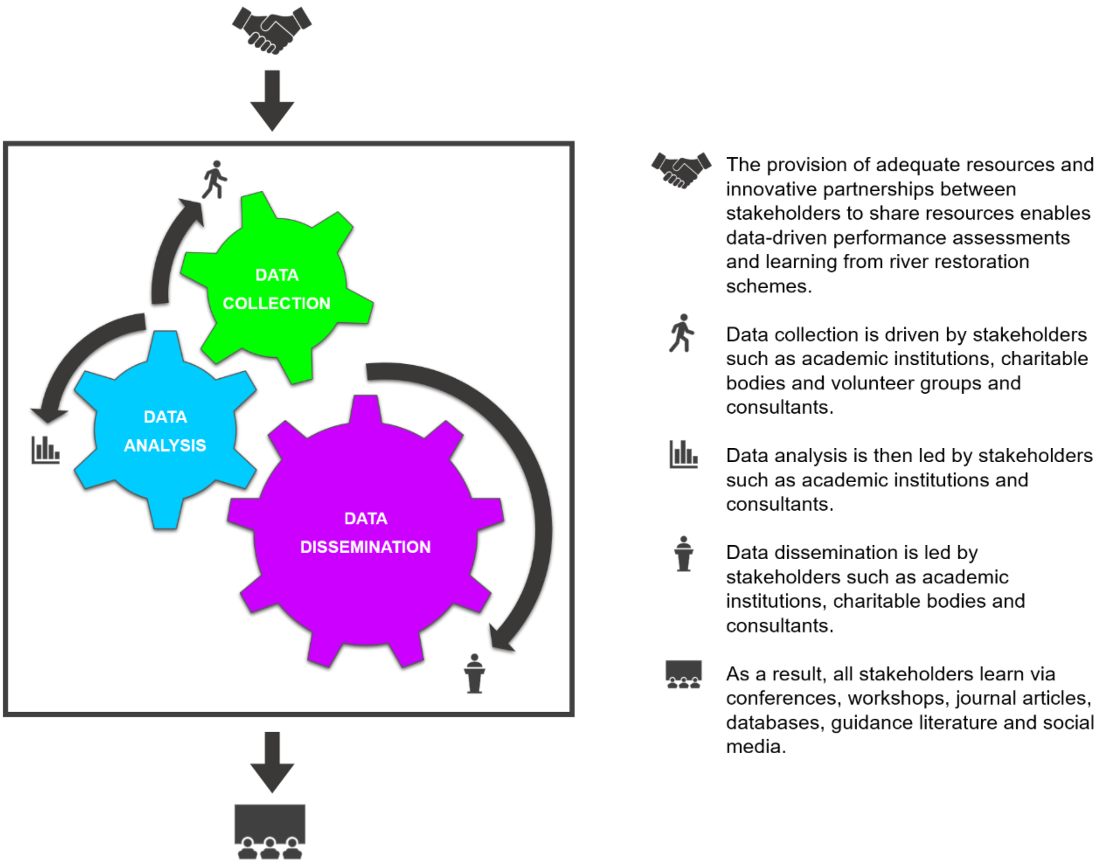


Figure 1.6 A conceptualisation of the integral components of an effective data-driven river restoration monitoring programme and the involvement of stakeholders within the process. Data collection, data analysis and data dissemination are integral components of this evaluative process, which must all work together to facilitate learning amongst stakeholders and adaptive management.

1.6 Research Objectives

Chapter 1 has introduced the context for this research study. The need for river restoration monitoring is clear and river restoration monitoring is lacking in both quantity and quality. Citizen science frameworks for river restoration monitoring show promise for improving the monitoring of lower risk river restoration projects. Higher risk projects that also have a high learning potential may require a data-driven monitoring programme to better understand the uncertainty around these schemes. However, the application of data-driven monitoring techniques beyond academic inquiry appears to be limited. This research study aims to improve the potential for learning and adaptive management within in river restoration practice. More specifically, this research aims to *explore how methods for cost-effective data collection, processing, analysing and communicating physical habitat datasets may be more widely applied in both the science and practice of river restoration*. Based on this exploration, the thesis aims to present guidance for data-driven river restoration monitoring to improve the application of data-driven performance assessments beyond academia. This guidance will be presented as complementary rather than a replacement for existing guidance. The specific objectives of this research study are;

1) *To evaluate existing methods of data collection and analysis of physical habitat variables used in related monitoring studies to identify novel technologies for river restoration monitoring.*

2) *To explore the practical use of a novel technology for collecting physical habitat variables in the context of a case-study river restoration scheme.*

3) *To assess the geomorphological and physical habitat performance of the case-study restoration scheme over space and time to learn from the scheme, as well as explore the practicality of analysing larger river restoration datasets.*

4) *Based on the experiences of undertaking a river restoration monitoring programme, to critically evaluate the practical application of routine data-driven river restoration monitoring.*

5) *To present guidance for practitioners and the concept of a dissemination framework to facilitate data-driven river restoration monitoring beyond academia.*

1.7 Thesis Structure

The thesis is structured around the above objectives. Each chapter begins with an introduction and ends with a summary of the key outcomes. An overall concluding chapter summarises the wider significance of these learning outcomes and opportunities for further research. Figure 1.9 outlines the thesis structure, Chapter 1 has outlined the broad research context for this study, highlighting the current need of this research. Key themes discussed here are explored more thoroughly through the thesis. Chapter 2 explores the principles of river restoration monitoring including the existing state of data collection, data analysis and data dissemination methods. The chapter also reviews existing methods of analysis of geomorphological change and physical habitats and explores some novel analyses from other scientific disciplines as potential options for analysing river restoration performance. The review of data dissemination methods suggests many outlets for communicating river restoration performance are available but may be underused. This review identifies the Acoustic Doppler Current Profiler (ADCP) as a novel technology for potential application in river restoration monitoring.

Chapter 3 introduces a river restoration scheme on the River Rother, West Sussex as a case-study to facilitate the completion of the research objectives (Figs. 1.7 and 1.8). The chapter contextualises the demonstration site, discussing the scheme within the catchment context, so that the geomorphological and physical habitat performance of the scheme may be effectively assessed in relation to baseline information later in the thesis. Chapter 4 outlines the details of the monitoring survey, post-processing protocol for the ADCP data and methods of analysis for evaluating physical habitat performance of the case-study. Chapter 5 and 6 present the geomorphological and physical habitat results of the 18 month monitoring programme of the case-study, respectively. These chapters identify key learning outcomes which may inform the future restoration of lowland sandy agricultural rivers and spawning habitat restoration schemes.

Figure 1.8 The River Rother, West Sussex prior to restoration in July 2013

Figure 1.7 The River Rother, West Sussex one year after restoration in August 2014

Chapter 7 reviews the use of data driven performance assessment for evaluating the performance of the case-study river restoration scheme. More specifically it explores the practicality of using the methods of data collection and analyses used within this thesis routinely within river restoration monitoring. The chapter presents a tool which may be complementary to existing guidance for justifying resources for data driven performance assessments. In addition, the chapter presents a framework which aims to provide guidance and a standardised method of disseminating high resolution river restoration monitoring relative to project objectives. The potential impact of the research is highlighted along with opportunities for further research beyond the scope of this research study in a concluding chapter, Chapter 8.

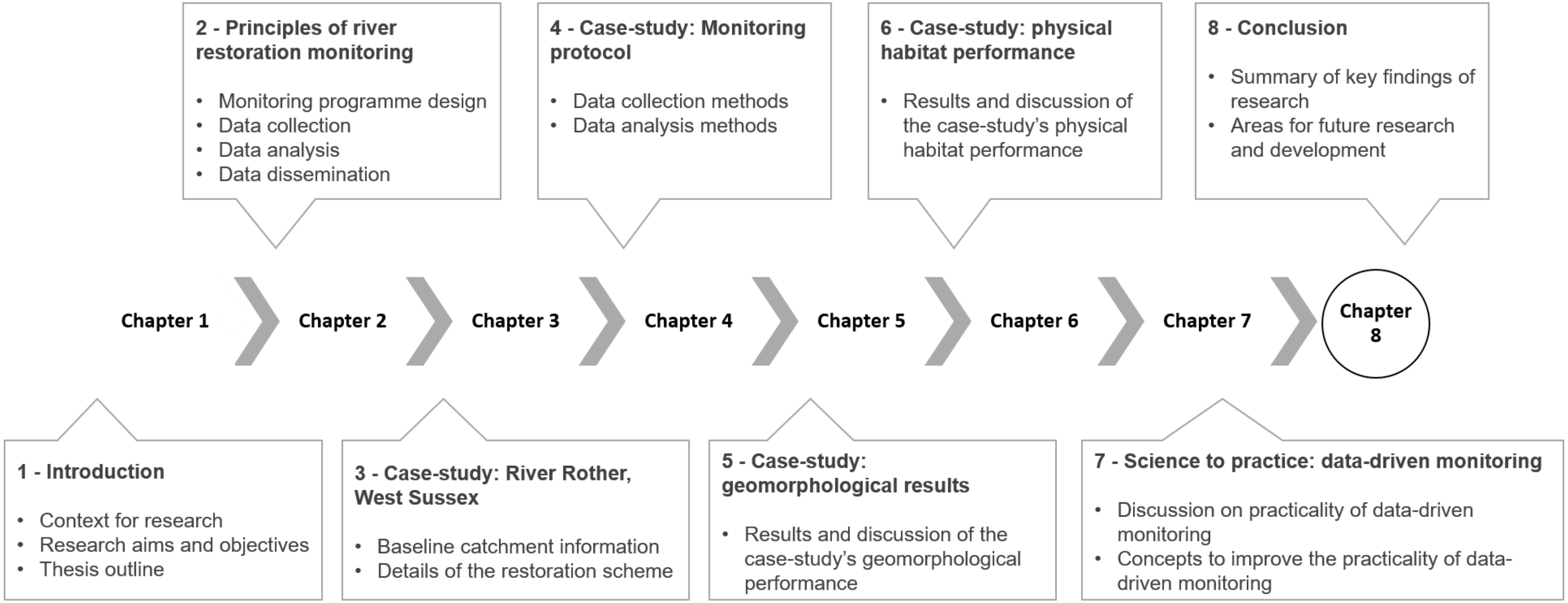


Figure 1.9 Thesis outline

2 Principles of River Restoration Monitoring

2.1 Introduction

This chapter reviews the literature and best-practice guidance on the principles of river restoration monitoring. It identifies areas for future research and informs the design of the monitoring programme used in the case-study of this thesis (Chapters 3 and 4). The structure of this chapter reflects the process of undertaking a river restoration monitoring programme (Figure 2.1). Typically monitoring may be viewed as a post-restoration activity, but it is an integral component of river restoration monitoring that should ideally be undertaken at least from the inception of the project (this may either be new project or a new phase of an existing project). It is important to design the monitoring programme so adequate baseline data can be collected to inform the setting of appropriate aims and objectives. From understanding the objectives of the project, appropriate data collection and analyses may be performed to measure the success of a scheme. A key component of learning and adaptive management within river restoration is the sharing of knowledge of both success and failures of a scheme. Therefore, the final component of the river restoration monitoring programme discussed within this chapter is a review of opportunities for data dissemination. The data collection and analyses sections are focused on geomorphological and physical habitat monitoring techniques, as this is the focus of this study (Chapter 1). However, the other sections are broader and may be applicable to a range of river restoration schemes.

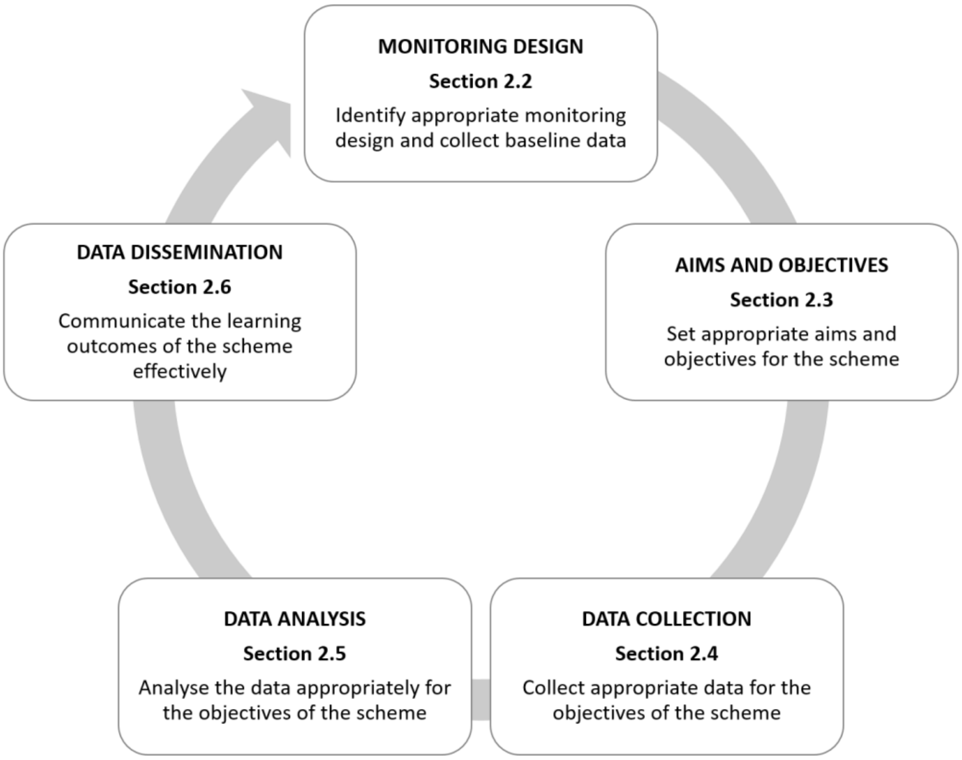


Figure 2.1 Structure of Chapter 2 reflecting the process of river restoration monitoring. This is a cyclical process ideally starting with baseline monitoring to maximise the opportunity for learning, but the process may be started throughout the cycle. The learning outcomes should encourage adaptive management and the process should restart armed with the newly gained knowledge.

2.2 Monitoring Programme Design

The design of a monitoring programme is integral to the success of learning from river restoration schemes, and (as discussed in section 1.3.1) limitations in monitoring programme design have been a significant barrier to river restoration monitoring (Vaughan et al., 2009; Buchanan et al., 2012; Gilvear, 2013, EA, 2015). This section will review three monitoring design frameworks in greater detail, namely, before and after (BA), before, after, control and impact (BACI), and multiple, before, after, control and impact (MBACI) assessments (Fig. 2.2). These frameworks are not dissimilar in their overall principle of detecting changes relative to the baseline conditions but they differ in the range of data that is collected. BACI and MBACI assessments offer more complex designs than the BA assessments, with the aim of collecting a wider range of data to reduce the uncertainty in interpreting stochastic environmental processes.

a)

	Before	**After**
Impact	✓	✓
Control	✗	✗
Control … *n*	✗	✗

b)

	Before	**After**
Impact	✓	✓
Control	✓	✓
Control … *n*	✗	✗

c)

	Before	**After**
Impact	✓	✓
Control	✓	✓
Control … *n*	✓	✓

Figure 2.2 Monitoring designs - a) Before and After (BA), b) Before, After, Control, and Impact (BACI) and c) Multiple, Before, After, Control and Impact MBACI.

The BA assessment, the simplest of these frameworks, involves monitoring the area or ‘impact’ site which is directly targeted by the restoration activities with a period of pre-restoration (baseline) monitoring and a period of post-restoration monitoring. The baseline monitoring should comprehensively capture the pre-restoration condition of the targeted ‘impact’ area prior to restoration, and set the scheme within the wider catchment context (Downs and Kondolf, 2002). The benefits of this initial investment in pre-project monitoring are threefold (Downs et al., 2011);

- to guide the project to identify the most pressing issues within the impact area, and therefore refine suitable project objectives;

- to guide project design through contributing to the development of analytical references where reference reaches are unavailable;
- to provide a benchmark from which change and ultimately performance can be measured when complemented with a suite of post-restoration surveys.

A comprehensive baseline survey dataset which satisfies the demands of the project objectives is considered the crux of this assessment process as it controls the ability to draw conclusions on project performance (England et al., 2008). Therefore, the learning potential from river restoration projects has been linked to the level of commitment to baseline data collection (Palmer et al., 2005).

The learning potential from river restoration monitoring typically increases with the length of the monitoring programme, as geomorphological and ecological adjustments may still be occurring decades following restoration (see Chapter 1, Downs and Kondolf, 2002; Woolsey et al., 2007). Post-restoration monitoring in the period immediately following restoration should be frequent and intense to capture the immediate response of the scheme (Kondolf and Micheli, 1995). Thereafter, the frequency of the post-restoration surveys can potentially decrease over time, as the river adjusts to a disturbance and continues along the path of its long-term trajectory (Grayson et al., 1999, Woolsey et al., 2007, Erwin et al., 2016). The best practice guidance in the UK for geomorphological monitoring suggests that data should be collected immediately following implementation, three months post-construction to capture immediate change, and thereafter annually and following high flow events to capture any further geomorphic change (EA, 2007).

The length of the monitoring protocol and the required frequency of surveys are both a function of the hydrological regime and sensitivity of the river system (Roni et al., 2005). For example, monitoring of the Provo River Restoration Project, Utah, US, identified rapid geomorphic change within the first two years post-restoration. In the following years, significant flow events were observed which contributed to further morphological change, however, this change occurred at a lower magnitude than was observed immediately post-restoration (Erwin et al., 2016). Therefore, in this example, the need for continued long-term monitoring is less pertinent as the resilience of the scheme has been tested by significant flow events. However, in other catchments around the world, particularly those of drier desert or Mediterranean climates, a more extended monitoring period may be required to test the resilience of restoration measures against significant flow events (Kondolf et al., 2007; Erwin et al., 2016)

The monitoring of a rehabilitation scheme on River Idle, Nottinghamshire, UK in 1996 exemplifies the BA approach. The River Idle was channelised for flood defence during the

1980s but was rehabilitated in 1996 using a range of techniques including flow deflectors to improve the ecological and aesthetic potential of the reach (Downs and Thorne, 1998; Downs and Thorne, 2000). The baseline survey revealed a uniform trapezoidal cross-sectional morphology throughout the reach with poor morphological diversity. The post-restoration survey indicated that the flow deflectors had promoted scour within the reach and improved the ecological potential by increasing the morphological complexity and exposing buried gravels (Skinner et al., 1999; Downs and Thorne, 2000). Hydraulic modelling indicated that the small losses in flood conveyance were sufficiently acceptable given the improvements (Downs and Thorne, 2000), thereby indicating that the scheme was successful in meeting project objectives in the short-term.

A subsequent survey was not undertaken until in 2010, and this indicated an improved morphological diversity compared to the immediate post-restoration survey (Soar and Downs, 2011). This indicates that the rehabilitation of the River Idle had been successful in the long term. However, further analysis indicated that the flow deflectors had potentially underperformed and morphological diversity was not maintained at higher flows (Soar and Downs, 2011; Soar, 2015). This example demonstrates the learning potential from monitoring with the BA approach, but given the lack of a control site, it is not clear to what extent the changes resulted as a direct result of intervention. The longer-term monitoring results (>10 years) yielded some valuable learning outcomes for restoration using flow deflectors and this example highlights the value of continued monitoring. Nonetheless, these learning outcomes are neither widely available nor publicised, thus demonstrating the issue of communication in river restoration monitoring as mentioned in Chapter 1.

The second and third design frameworks, namely the BACI and MBACI assessments (Figure 2.2), can reduce some of the uncertainty associated with environmental noise by incorporating at least one additional control site (Green, 1979; Chessman and Jones, 2001; Summers et al., 2015). A control site must exhibit similar forms and processes, and prevailing environmental conditions to the target (impact) reach (Downes et al., 2002). Unlike laboratory controls, control sites are still subject to the same processes as the impact site, therefore, they allow the relative role of the restoration measure to be determined in comparison to prevailing environmental processes (Downes et al., 2002). Both the impact and control reaches are routinely monitored post-restoration at either specified time points or at a range of flow events for a set time-period. The BACI monitoring design helps to eliminate real trends from environmental noise (Summers et al., 2015) and the MBACI may reduce uncertainty surrounding the effects of environmental noise even further through multiple control sites.

The monitoring programme of the restoration of the Williams River, New South Wales, Australia, utilised the BACI design. The scheme introduced engineered log jams (ELJ) to meet several restoration objectives; one of these objectives was to reintroduce morphological diversity that had been lost through vegetation clearance and catchment deforestation (Brookes et al., 2004). In the year following construction, three flood events (with a magnitude greater than the mean annual flood) were experienced, and after a year, a topographic resurvey indicated an improved morphological diversity in the restored reach (Brookes et al., 2004). However, a control reach which was surveyed also exhibited increased morphological diversity although not to the same extent as the impact reach (Brookes et al., 2004). In this example, although morphological diversity increased, the use of a control site was effective in identifying that some changes detected in the restored reach would probably have occurred within the monitoring period without direct intervention. Therefore, in this example, an MBACI monitoring design may have helped identify the extent of the change that could have occurred without direct intervention.

Whilst the BACI assessment has been demonstrated as useful in supporting the learning from river restoration, a BACI or MBACI assessment may not always be possible or realistic. Control reaches which replicate the same processes observed at the impact site can be difficult to identify as processes in river systems are complex (Summers et al., 2015). Furthermore, the use of a control site may not always be appropriate, particularly if the restoration activities have the potential to induce change throughout the system, for example dam removal (Kibler et al., 2011). The use of multiple control sites may be useful in exploring the stochasticity of environmental processes between sites (Summers et al., 2015). However, the use of a BACI or MBACI assessments are likely to have higher associated costs and, as discussed in Chapter 1, the monitoring of many river restoration schemes is financially limited. This may partially explain why Morandi et al., (2014) observed the BA assessment as the most common form of monitoring design when reviewing French practices.

2.3 Setting Restoration Aims and Objectives

Objective setting is a critical part of the river restoration process to guide restoration approaches but also provide rigorous performance criteria for evaluation. As such, this should be informed by a period of baseline monitoring and consideration of wider catchment processes (Downs et al., 2011). As discussed in Chapter 1.3, a complete lack of objective setting or objectives which lack quantifiable elements are a major limitation of many restoration schemes (England et al., 2008; Brierley et al., 2010). Furthermore, many river restoration schemes have been unsuccessful as they have not set objectives appropriate to the catchment conditions (Roni et al., 2008; Friberg et al., 2016). River restoration objectives should be "*specific and measurable achievements that are necessary to reach a restoration goal*", in practice, however, many objectives reflect the definition of river restoration aims, namely, "*broadly stated aims or desired outcomes of restoration effort, including the main biological outcome to be achieved*" (Roni and Beechie, 2012). The guidance for practitioners advocates the use of SMART objectives (Doran, 1981, Chapter 1) to improve the rigour of target setting (Woodward and Hollar, 2011; RRC, 2011).

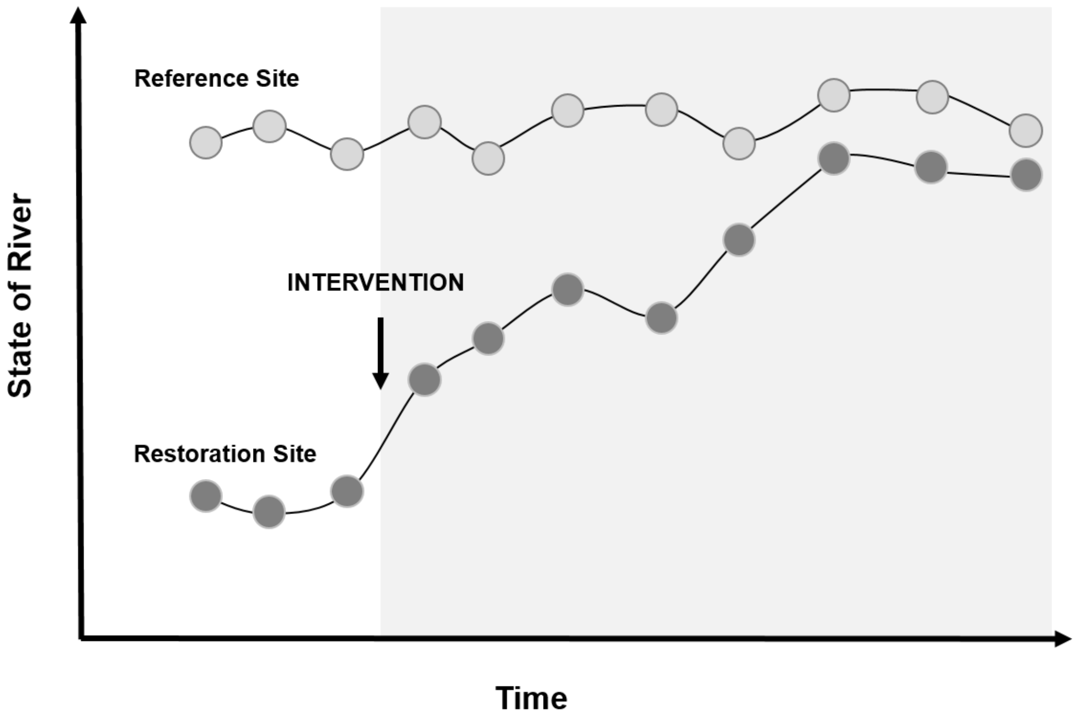


Figure 2.3 Moving targets. The target (circle on the reference site line) is based on the performance of the reference reach so changes over time with fluctuating environmental conditions.

There is a growing body of literature which has called for ‘moving targets’ in river restoration, which account for uncertainty in restoration, system change and future trajectories (Heirs et al., 2012; Hughes et al., 2012; Perring et al., 2015; Brierley and Friars, 2015; Maier et al., 2016). The moving target approach proposes to incorporate inherent variability within the system through simultaneously analysing the conditions of a reference reach within the monitoring programme (Heirs et al., 2012). A reference reach is a ‘blueprint’ for river restoration design, a reach that is subject to very similar processes to the restoration site but exhibits desirable characteristics (Rosgen, 1998). The observation of natural variation within reference reaches can help identify potential trajectories of the restoration scheme and therefore, identify realistic targets (Brierley and Friars, 2015). As the reference reach is subject to the same prevailing environmental processes and the impact reach, the target may change over time (Fig. 2.3).

The use of moving targets acknowledges that there is no static end-point to restoration but rather that the reach has the potential to evolve towards more than one state dependent on the future state of environmental processes (Heirs et al., 2012; Hughes et al., 2012; Brierley and Friars, 2015; Maier et al., 2016). Additionally, the use of moving targets acknowledges that the outcomes of a restoration scheme may improve incrementally over time particularly as further restoration work is performed within the catchment. Whilst this is a progressive approach, there are presently very few worked examples which detail how this approach can be applied in practice and replicated.

A potential limitation of this approach is that in many river systems around the world, channel modification and or wider catchment modification has been so extensive, that it may be difficult or even impossible to identify a reference reach (Downs et al., 2011; Hughes et al., 2012). Additionally, the binary nature of target setting in general (i.e. success or failure) may also be a limitation. This approach may fuel a reluctance to monitor schemes, whereas a non-binary approach that might make recommendations could potentially be more supportive and encouraging in an adaptive management situation. A non-binary approach may be useful if a reference condition is unavailable and uncertainty around ‘success’ may be greater.

2.4 Data Collection

The need for monitoring of river restoration schemes is irrefutable (Chapter 1.3). The monitoring of every river restoration scheme, particularly using data-driven techniques, is an unrealistic target. A range of techniques are available to collect data to support the assessment of physical habitat restoration performance so that lessons can be learnt from all schemes. Guidance typically indicates that resources should be targeted at higher risk projects which are likely to yield the most valuable learning experiences. There are several decision support frameworks (DSF) available to help assess the risk of a restoration project and choose the most appropriate method of assessment (e.g. England et al., 2008, RRC 2011, Thorne et al., 2015).

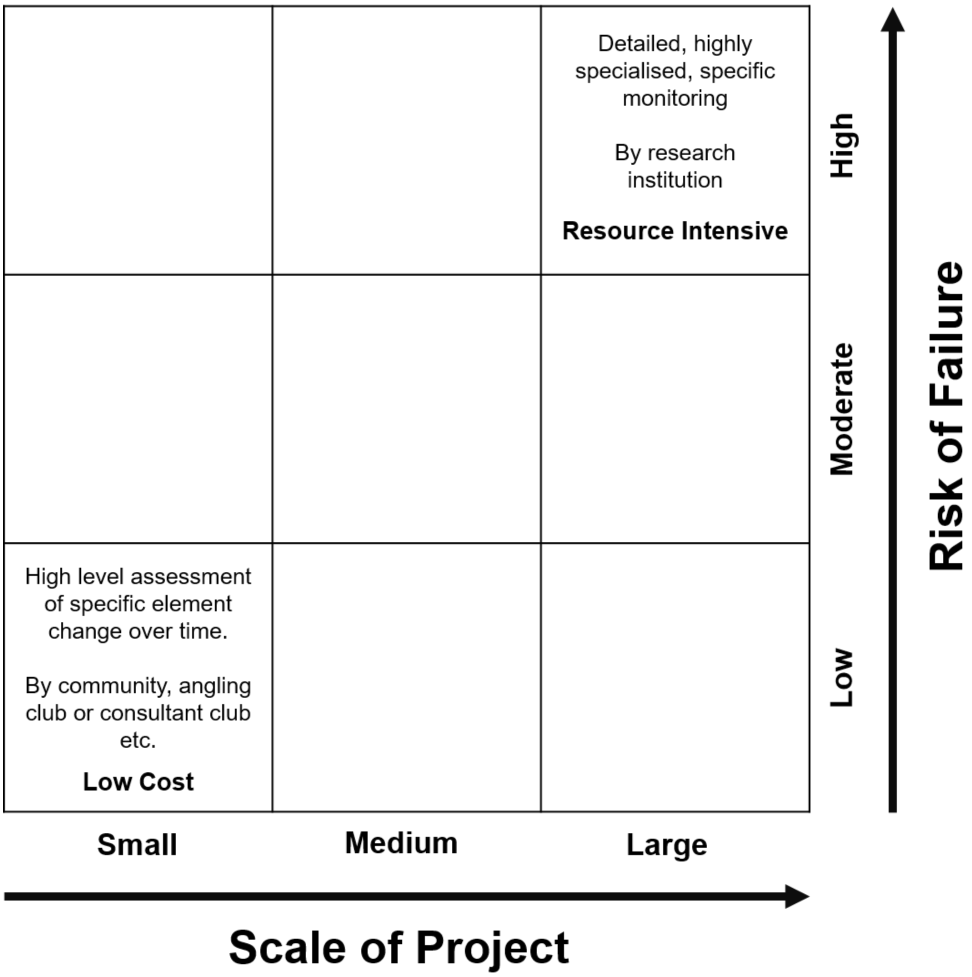


Figure 2.4 Example of matrix adapted from the RRC PRAGMO guidance (2011). This matrix considers the scale of the project and project's risk of failure. A high-risk project is defined as 'a project involving a new technique, suite of techniques or using an established technique in a new environment e.g. the use of large woody debris in an urban environment, and a low-risk project is defined as 'a project involving established techniques e.g. riffle creation in a lowland chalk catchment.

Guidance on targeting resources suggests that the monitoring of lower risk restoration projects would typically be led by the local community or interest clubs (e.g. angling clubs). In contrast, high-risk projects would typically be led by research institutions due to the specific knowledge and expertise required for such assessments (Fig. 2.4, RRC, 2011). Methods of geomorphological monitoring at the lower scales of risk include fixed-point photography and rapid visual surveys of surface flow types (RRC, 2011; 2016). However, the existing guidance does not consider the profile of schemes or potential breadth of learning by stakeholders when targeting resources. There may be a case for schemes that have a higher profile or a higher level of stakeholder engagement warranting more resources for monitoring. Nor does the existing guidance widely consider the advent of rapid data collection technologies, which may be used to either collect or collate data.

There are a multitude of rapid habitat assessment protocols which have been developed in response to legislative drivers aiming to improve river conditions, such as the EU WFD and US CWA (Fernandez et al., 2011; Belletti et al., 2015). These assessment protocols were initially developed to identify and prioritise areas for intervention. However, there has been a more recent move towards utilising these techniques for river restoration monitoring (Fryirs, 2015). Examples of these include the River Habitat Survey (RHS, Raven et al., 1997; Raven et al., 2000), GeoRHS (Branson et al., 2005), Urban River Survey (Shuker et al., 2015), AusRivAs (Parsons et al., 2004) and MoRPh (Shuker et al., 2017).

These methods require nominal training and are based on minimally invasive measurements and visual assessment, consequently, these types of assessment methods are ideal for large scale cost-effective implementation (Harvey et al., 2008; Zavadil et al., 2012; Shuker et al., 2015). The nature of these datasets has been useful in identifying broad-scale geomorphological trends to inform adaptive management (Emery et al., 2004; Smith et al., 2014; Naura et al., 2016). These assessment methods may also help to meet the recommendations of the ICUN report (Addy et al., 2016) on gathering evidence and project evaluation using cost-effective citizen science based approaches (Shuker et al., 2017). Whilst this may be cost-effective when using volunteers, if an expert is required to provide an improved level of accuracy, particularly for longer reaches, this method may still be resource intensive (Zavadil et al., 2012; Bentley et al., 2016).

A disadvantage of these assessment methods is that they can potentially oversimplify the complexity of physical processes (Clifford et al., 2006; Harvey et al., 2008; Belletti et al., 2015; Woodget et al., 2016). Researchers have shown that physical biotope complexity may be significantly underestimated when mapped from the bankside as opposed to being defined using high resolution data or aerial imagery (Milan et al., 2010; Woodget et al., 2016; Bentley et al., 2016). As these rapid assessment surveys are dependent on expert judgement, they are criticised for lacking objectivity. Particularly in the situations of inter-reach comparisons and long-term monitoring, several 'experts' may undertake monitoring of the same site which could potentially introduce inconsistency into assessments (Maddock, 1999). The ICUN report also recommended improvements to objectivity in project evaluations (Addy et al., 2016), consequently rapid evaluation techniques are not a universal solution to river restoration monitoring.

For higher risk projects, additional data collection of topographic and cross-sectional data is advised to capture geomorphological and physical habitat change in more detail (RRC, 2011). Traditionally these datasets have been captured in cross-sections, using a total station or GPS to capture depth/elevation data whilst point velocity measurements have been estimated using an Electromagnetic Current Meters (ECMs) (Emery et al, 2003; Wallis et al., 2012) or Acoustic Doppler Velocimeters (ADVs) (Wilcox and Wohl, 2007; Wilcox et al., 2011, Cockburn et al., 2015). However, these data collection methods have been associated with resource-intensive monitoring campaigns. Monitoring guidance suggests that for a ~10m wide channel, approximately 10-15 cross-sections can be captured during a day of surveying (EA, 2007). In comparison to more advanced technologies, such as remote sensing, this yields low resolution data. Additionally, capturing cross-sectional measurements has often been parsimonious to save resources, at the expense of data resolution (Maddock, 1999).

With limited access to high-resolution velocity data, hydraulic models have been used to quantify physical habitat (Crowder and Diplas, 2002; Brown and Pasternack, 2009; Wheaton et al., 2010; Pasternack and Brown, 2013; Tamminga et al., 2015). In practice, modelling has been previously viewed as the 'only' method for understanding complex environmental processes (Inkpen and Wilson, 2013). Gostner (2012) argued the benefits of numerical modelling of river channels over fieldwork were three-fold; firstly, the modelled data is spatially continuous, whereas 'real' data (even of the highest resolution) requires interpolation as the spatial distribution of the data is determined by the equipment used. Secondly, modelled data is free from bias introduced to the dataset from sampling procedure in the data (i.e. operator variability). However, Gostner (2012) did not appear to appreciate that models may use topographic data that has potentially been subject to operator variability. Finally, velocity can be simulated under a range of flow conditions allowing the investigator to set the discharge, rather than discharge being a pre-determined environmental variable during data collection that may later limit the analysis of physical habitat.

In addition to these three benefits, hydraulic modelling can reduce the risks associated with data collection for physical habitat assessments, particularly when coupled with remotely sensed data (Lane and Carbonneau, 2008). However, a significant caveat with numerical modelling is that the data produced is a representation of reality that is limited by the ability of empirical equations to replicate the inherent stochasticity of environmental systems. Furthermore, in smaller rivers and streams, small topographical errors can result in variations in velocity and thus physical models (Legleiter et al., 2011; Legleiter, 2012). As river restoration is undertaken at a range of scales but often at the smaller scale and opportunistically (Bruce-Burgess, 2004), hydraulic modelling may not be preferable to the collection of real data.

2.4.1 Advanced Technologies

Since the early 2000s, the advancement of technology and computer processing presents an increasing opportunity to work with ‘real’ rather than simulated data in the river sciences (Piegay et al., 2015). This pattern of change has been compared to transformations in other physical sciences in the early 20th century through the emergence of new technologies (Gregory et al., 2014; Viles, 2016). Fluvial geomorphology is undergoing a methodological evolution which has influenced the nature of geomorphological studies (Wohl, 2014; Viles, 2016). In the mid-20th Century, geomorphological studies mostly focused on advancing conceptual frameworks, such as work on river channel patterns by Leopold and Wolman (1957). However, more recently the focus has shifted towards data-rich exploratory research (Wohl, 2014). The advancement of technology has widened the scope and scale of data collection to explore fluvial processes (e.g. Vericat et al., 2014; Leyland et al., 2015; Hackney et al., 2015) to the extent where extra-terrestrial applications of fluvial geomorphology are emerging (Baker et al., 2015; Auld and Dixon, 2016).

Remote sensing methods have been widely used to collect physical habitat data and detect geomorphological change. These include optical remote sensing (Fonstad and Marcus, 2005; Marcus and Fonstad, 2008; Bentley et al., 2016), laser scanning (Heritage and Hetherington, 2007; Hall et al., 2009; Milan et al., 2010) and structure-from-motion photogrammetry (Woodget et al., 2015a; 2016; Tamminga et al., 2015). They have the potential to improve the spatial resolution and extent of data collection in river restoration monitoring programmes (Lane and Carbonneau, 2008; Gilvear et al., 2016). The increasing availability of free aerial imagery from virtual globes such as Google Earth (Bentley et al., 2016) and low-cost aerial imagery from Unmanned Aerial Vehicles (UAV) (Woodget et al., 2016; Rivas-Cassado et al., 2016) are improving the ability to delineate physical biotopes. These resources present an opportunity to improve the rigour of physical biotopes monitoring methods (which would be highly beneficial for lower risk projects), as mapping these features from above is more accurate than when mapped from the bank side (Woodget et al., 2016).

Higher resolution data collection using some remote sensing methods such as aerial-borne LiDAR can be resource intensive. However, the application of structure-from-motion (SFM) to river environments using UAVs and other personal technologies has sparked excitement as a low-cost alternative to LiDAR (Westoby et al., 2012, Fonstad et al., 2013; Michelletti et al., 2015). Recently, the use of UAV platforms has been applied to quantify physical habitat attributes at the reach scale including topography and substrate (Woodget et al., 2015a; 2015b; 2016). A survey of the River Arrow, Warwickshire, indicated that using the UAV-SFM approach could quantify topography at a 0.02m resolution of a reach of up to 200m in length and 40m wide within one day (Woodget et al., 2015a). This example demonstrates the potential for improved expediency and data resolution. This may potentially reduce financial barriers to data-driven river restoration monitoring. However, some substantial post-processing of the data may be required (Woodget et al., 2015a, Rivas-Cassado et al., 2016; Marteau et al., 2016), which could possibly add to the cost of this type of survey.

As with many photogrammetric techniques, SFM and LiDAR are limited in the environments to which they can be applied, notably in deep and or turbid environments (Legleiter et al., 2012; Woodget et al., 2015). The optimum detection level has been noted in water depths less than 0.2m but may provide suitable measurements in water depths up to 0.7 m in good conditions (i.e. lighting and low surface roughness) (Woodget et al. (2015). LiDAR has been observed to have significantly deeper maximum detection level of 10m water depth but a minimum detection level of 0.5m water depth (Milan and Heritage, 2012). In a review of the UK National River Restoration Inventory, lowland sites were more likely to be restored (Smith et al., 2014). These sites may be more commonly associated with finer sediments and turbid water in which SFM and LiDAR may not be an effective tool. Consequently, the range of environments to which these technologies can be applied for river restoration monitoring programmes is potentially limited.

In addition, weather may affect both data quality and the ability to operate the equipment (Westoby et al., 2012; Marteau et al., 2016; Woodget et al., 2017). Consequently, the application of these technologies throughout the year to capture river restoration monitoring data may also be limited, as weather, flows and suspended sediment transport (and thus turbidity) may fluctuate. The need to capture physical habitats throughout variable conditions is critical as physical habitat spaces are dynamic over space and time (Wallis et al., 2012). Therefore, further exploration of the validity of using multiple technologies to collect river restoration monitoring data over time would be useful.

Increasingly, these photogrammetric approaches are being used to capture geomorphological and physical habitat variables other than elevation. Until recently, photogrammetric approaches were largely used to quantifying topography and coupled with hydraulic modelling to quantify velocity (Tamminga et al., 2015). The increased topographic resolution may reduce the risk of error associated with the modelling process.

However, more recently UAVs have been used for airborne velocimetry surveys which could provide an alternative to hydraulic modelling in deriving spatially continuous velocity data (Detert and Wietbrecht, 2015; Bolognesi et al., 2016; Tauro et al., 2016; Detert et al., 2017). In addition, grain size (also an important physical habitat parameter) has been quantified using SFM (Woodget et al., 2017). Although these techniques are still largely at the proof of concept stage, it presents an exciting advancement for river restoration and demonstrates the rapid pace of technology development. However, given the limitations of using UAVs listed above, it is likely that this technique may need to be used in tandem with other technologies.

2.4.2 Acoustic Doppler Current Profiler

In contrast to the photogrammetric data collection methods (such as SFM and LiDAR), acoustic technology emits sound from a transducer and utilises the return signal from particulate matter within the water column to estimate velocity (Kostaschuk et al., 2005). Acoustic Doppler Current Profilers (ADCP) can operate in a wider range of in-channel conditions and could be considered as an advantageous tool for river restoration monitoring, particularly in reaches which are turbid or deep. There are different types of ADCP, namely stationary and moving-boat ADCPs (Muste et al., 2004a; 2004b). Stationary ADCPs sample velocity measurements from a fixed location, commonly either vertically (looking upwards) from the stream bed, or horizontally (looking across the channel). Moving-boat ADCPs are downwards facing and are typically mounted on mobile platforms (e.g. float, kayak, boat). Moving boat ADCPs sample vertically through the water column but in locations defined by the operator. This study is primarily interested in the potential of moving boat ADCPs for river restoration monitoring, therefore, hereafter the term ADCP will refer to moving boat ADCPs. As the ADCP is mounted to a float its application also has limitations, particularly in shallow or boulder streams as the float needs to be able to freely move. However, operational issues (particularly in the UK) may be less of an issue as restoration is more common in lowland streams which are typically wider and deeper (Smith et al., 2013).

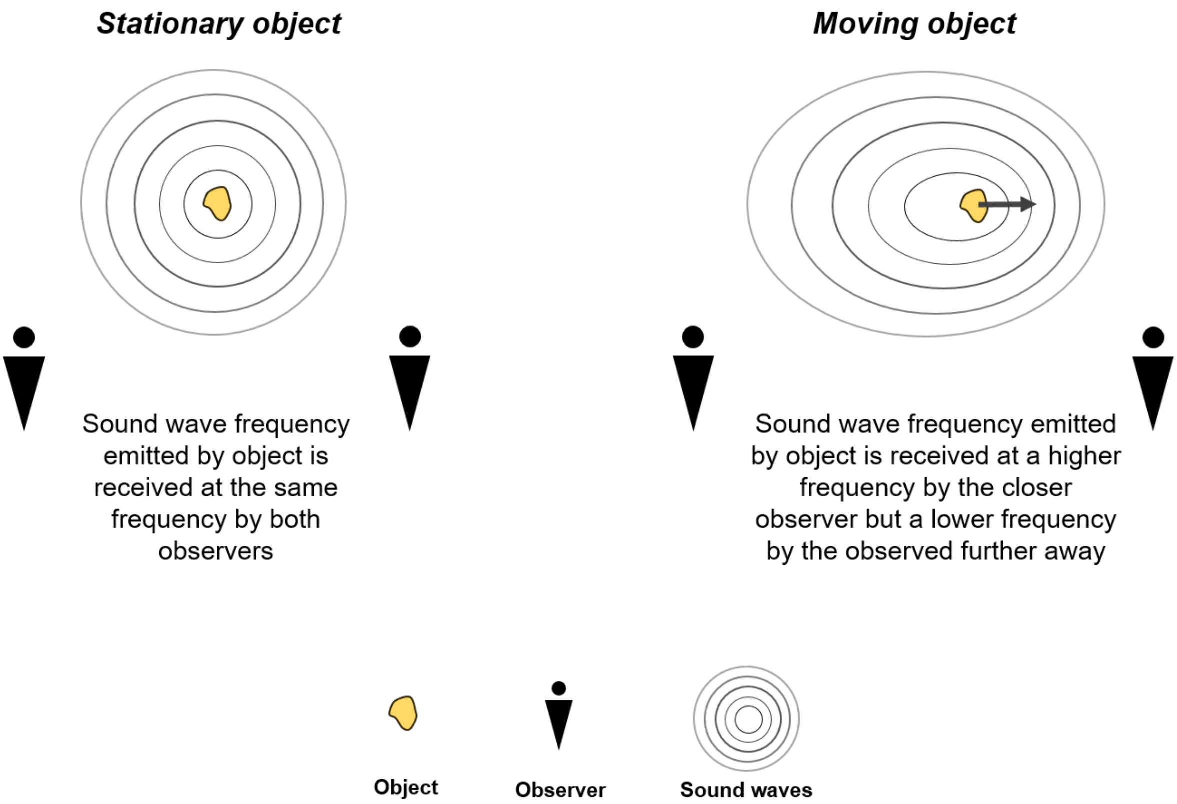


Figure 2.5 The Doppler Effect - the effect of a moving object on sound frequency

The ADCP calculates velocity using the principle of the Doppler shift, where changes in sound wave frequency are observed through the relative motion between the source and the observer (see Kostaschuk et al., 2005). If an object is moving away from an observer, the frequency of the sound waves decrease but if an object is moving towards an observer, the frequency of the sound wave increases (Fig. 2.5). The ADCP operates through pulsing sound through the water column from a transducer at a known frequency, and sound waves are reflected by particulate matter within the water column. The transducer receives some of this backscatter and calculates the velocity using the change in frequency emitted and received by the transducer. An ADCP can quantify flow in three dimensions at multiple depths within the water column and uses a separate beam to quantify each dimension. Many ADCP traducers have four acoustic beams arranged in a Janus configuration (Figure 2.6) with the fourth used for calculating an error velocity, however, this design is not exclusive (Mueller and Wagner, 2009). As a result of the Janus configuration, where the two pairs of beams are angled to look in opposite directions, the sampling area of velocity increases with depth (Fig. 2.6). The ADCP assumes flow is homogeneous within the sample area, but the risk for error in velocity measurement increases with depth (Muste et al., 2004a). The ADCP is also frequently coupled with an echo sounder to provide a simultaneous estimation of water depth.

The ADCP has become an established technology for quantifying open-channel discharge

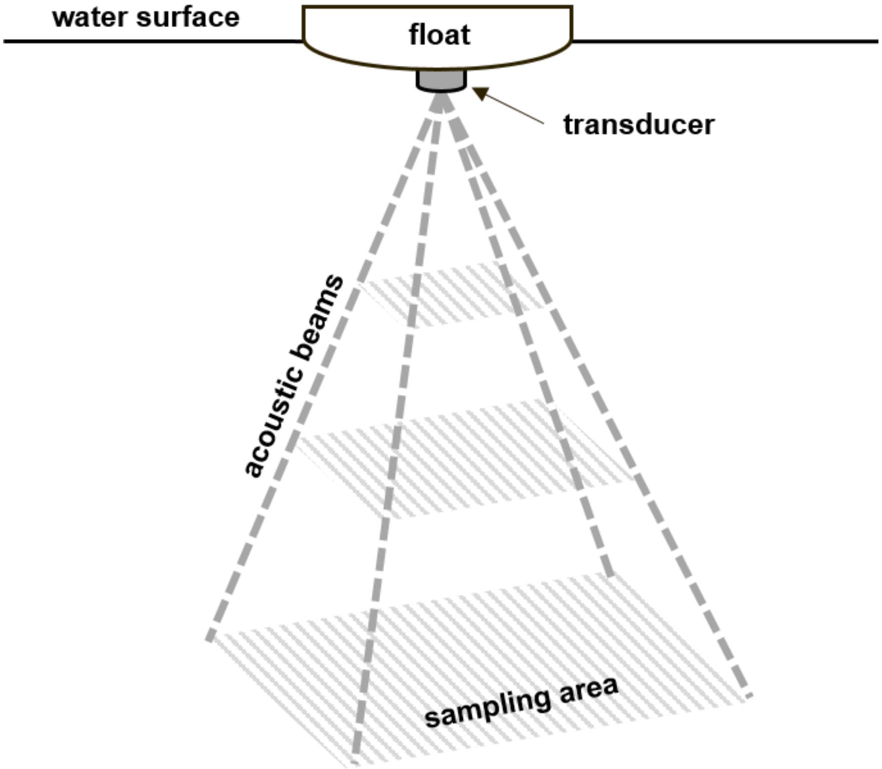


Figure 2.6 Principles of operation of an ADCP. The angling of the beams increases the sampling area with depth.

measurements as it can simultaneously measure velocity and depth more efficiently than point measurement techniques (Stone and Hotchkiss, 2007). It is used widely for routine stream gauging and calibration of hydraulic models by governmental agencies for example, the Environment Agency (UK), the United States Geological Survey (USGS) and the Water Survey of Canada (Muste et al., 2004a; Mueller and Wagner, 2009, Fox et al., 2016). The ADCP has been found comparable in accuracy to point measurement techniques such as the ADV (Stone and Hotchkiss, 2007; Gunawan et al., 2008; 2010), although operational issues in very shallow and vegetated streams are acknowledged (Bialik et al., 2014). The ADCP has become such a standard piece of technology that it has been used to verify the accuracy of novel velocimetry methods (Detert et al., 2017).

The ADCP has also been widely used beyond stream gauging for a range scientific investigations. These have included hydraulic model calibration (Williams et al., 2013), the calculation of shear stress (Sime et al., 2007), quantification of sediment transport (Dinehart and Burau, 2005; Jamieson et al., 2011; Rennie et al; 2017), mapping physical habitats (Milan and Heritage, 2012), the quantification of hydraulic properties of in-channel structures (Kriechbaumer et al., 2016) and classifying substrate (Sheilds, 2009a, 2009b). ADCP data can be visualised in a continuous spatial framework, examples of this include the quantification of flow and morphology around in channel structures (Jamieson et al., 2011; Jamieson et al., 2013), mapping bedload transport in a braided river (Williams et al., 2015) and mapping physical biotopes (Milan and Heritage, 2012). The ADCP has also been used in ecological studies, for example, Marchildon et al (2010) used an ADCP to evaluate redd (spawning nest) selection of brown trout.

The ADCP has been recommended as an alternative to SFM for river restoration monitoring where the application of this photogrammetric technique may not be feasible (Marteau et al., 2016). However, there appears to be little demonstrable evidence of the ADCP being previously applied for river restoration monitoring, despite it being a well-established technology within both science and practice. Observations from Gunawan et al. (2010) have suggested that data processing to extract velocity data (beyond that required by the standard discharge processing software used in industry) could be laborious. Many of the guidance reports for river restoration monitoring do not provide guidance on the use of advanced technologies such as ADCPs for river restoration monitoring. This is possibly because these technologies are relatively new but also may be because data-driven river restoration monitoring is typically associated with research institutions. Consequently, the later chapters of this thesis will explore the practical use of the ADCP for providing a data-driven performance assessment for a river restoration scheme.

2.5 Data Analysis

It is clear from Section 2.4 that the ability to collect physical habitat data is vastly improving in both resolution and efficiency through technological developments. These improvements present an exciting opportunity to learn more about river environments and the effects of restoration. Guidance on choosing and performing appropriate data analysis is in general seen as a barrier within river restoration monitoring (England et al., 2008). However, the guidance for practitioners on how to analyse the increasing volume of data is limited even more so at present as technology has developed so rapidly. Some online guidance manuals are improving the visibility of geomorphological techniques, for example Cook et al. (2017) which provides additional detail on techniques that have been used in peer-reviewed journals. This resource is developing and more accessible than a peer-reviewed journal article but still appears to be geared more towards an academic audience. Clear guidance on how these techniques could be applied practically to physical habitat monitoring and river restoration monitoring in general could be highly beneficial. This section will review some methods that have been used to analyse geomorphological change and physical habitat performance.

2.5.1 Geomorphological Change and Diversity

Geomorphological adjustment is to be largely expected in any river, particularly following an intervention. Monitoring changes in channel bed, bank and floodplain elevations can identify how the river is adjusting over time in response to river restoration and if this aligns with project objectives. A typical approach to capture this is through the collection of cross-sections or more spatially continuous elevation data (Section 2.4). Simple statistical analyses may be performed on cross-sections to identify changes to channel geometry. For example, the standard deviation of channel widths was assessed from cross-sections taken on the River Rother following the reconnection of Shopham Loop in 2004. Between 2004 and 2009, the channel adjustments suggested that the channel was deepening following restoration (EA, 2015a). The standard deviation of depths in this monitoring study was used to indicate an improvement in the diversity of habitats (Holloway and Mant, 2011). An analysis of the distribution of depths (i.e. Skewness and Kurtosis) indicated that the channel was imitating a more natural form over time (Holloway and Mant, 2011).

In another example, the standard deviation of bankfull width and depth was also assessed from cross-sections taken on the River Wylye, Wiltshire following the restoration of Seven Hatches. This analysis indicated that the geomorphological diversity of the channel improved following restoration which was an objective of the scheme (EA, 2015b). However, simple statistical measures such as standard deviation of channel parameters may have the potential to overlook variability in cross-sectional form and complexity. Their use in conjunction with complexity measures such as length of the cross-section (Holloway and Mant, 2011) or cross-sectional asymmetry may identify more specific changes in channel form (Knighton, 1981; Rayberg and Neave, 2008; Neave and Rayberg, 2016). The analyses performed within these two studies were undertaken in MS Excel. These examples serve to highlight that even with relatively low-resolution cross-sectional data and limited resources, simple analyses can be performed to assess project performance relative to their objectives.

These spatial statistics may be applied to larger higher resolution datasets, along with other surface complexity metrics such as rugosity and coefficient of variation of geomorphological variables (Scown and Thoms, 2015). There are a range of complexity metrics available, but none appear to have become widely-accepted as a standard (Wohl, 2016). For larger data sets the use of geostatistics has been coupled with both variograms (Legleiter, 2014a; 2014b) and moving window analyses (Scown and Thoms, 2015) to assess the spatial organisation of these metrics. Interestingly, an investigation of floodplain morphology that performed a circular moving-window analysis of spatial complexity metrics, found the metrics were influenced by the size of the window (Scown and Thoms, 2015). Consequently, the ability to compare the results of river restoration schemes may be influenced by both the metric used and the scale over which it is assessed.

With repeat surveys of spatially continuous or high-resolution data that has been interpolated to form a Digital Elevation Model (DEM), it is also possible to analyse spatial topographic change and estimate sediment budgets. This may be achieved by subtracting the old DEM from the new DEM to give an estimate of volumetric change, this technique is known as the DEM of Difference (DoD). The DoD technique has been used widely to investigate geomorphological change in fluvial systems (see Williams, 2012 for review) and selectively but increasingly applied to quantify change from river restoration schemes (Wheaton et al., 2010b; Addy and Wilkinson, 2016; Norman et al., 2017).

For example, Wheaton et al. (2010b) used the DoD technique to evaluate the effectiveness of spawning habitat rehabilitation schemes downstream of a dam on the lower Mokelum River, California. Initial analysis of this scheme using the DoD technique highlighted that the reach which was restored saw a net loss of material following restoration. This indicated that seeded gravels were being lost from the reach. However, further spatial analysis of the DoD suggested that sediment was lost from the pools within the reach and not the area that was restored. This example highlights the value of the technique but also the specialist knowledge that is needed to critically evaluate the results of river restoration monitoring. Accompanying this research, the authors published their analytical tools for others to use and this was later developed into an ArcGIS plug-in (Wheaton et al., 2010a). This is a good example of researchers disseminating their work but the tools often require licences of expensive and less standard software.

2.5.2 Habitat Heterogeneity

Habitat heterogeneity is assumed to be associated with good ecological performance, despite contention within the literature (Chapter 1.2). Studies have quantified habitat heterogeneity using a variety of different methods and variables. Brooks et al. (2002) assessed heterogeneity using an analysis of variance (ANOVA) statistical test of sediment size, mean velocity, mean turbulent kinetic energy and spatial variation in turbulence variables. Harrision et al. (2004) also used an ANOVA test to assess habitat heterogeneity when comparing macroinvertebrate assemblages to average velocity, depth and substrate size. In contrast, Gostner et al. (2013) used a more simplistic approach to quantify habitat heterogeneity through calculating the coefficient of variation (CV) of velocity and depth. Their study used these calculations to present a hydromorphological index of diversity (HMID) that could be used to quantify the physical habitat performance of different restoration designs.

The index value ranges presented by Gostner et al. (2013) were based on testing within alpine streams and therefore, may require further calibration to be applied to other environments. Similar studies have since adopted the CV of velocity and depth to evaluate habitat heterogeneity (e.g. Massey et al., 2017; Holzapfel et al., 2017). Massey et al. (2017) noted a lack of correspondence between the HMID and the qualitative habitat evaluation index (a visual assessment method) in some environments. This could present issues when comparing the performance of river restoration schemes that have been evaluated using different methods and highlights the value of using multiple assessment methods.

In other approaches, the identification of distinct physical habitat patches has been used to assess habitat heterogeneity. This is a method that transcends both qualitative and quantitative physical habitat assessment methods. Qualitative methods of delineation are mostly conducted during walk-over surveys, for example, using surface flow types to identify a suite of physical biotopes which have ecological significance (Section 2.4). The increasing ability to collect hydraulic and topographic datasets allows a more objective quantitative approach to patch delineation. The use of cluster analysis for delineating physical habitats has been applied with different studies adopting different algorithms (Emery et al., 2003, Wallis et al, 2012), and also within the wider field of fluvial geomorphology (e.g. Nelson et al. 2014). Cluster Analysis is a method for categorising the data into sub-sections i.e. clusters, a universal principle of clustering algorithms is to maximise homogeneity within the clusters and heterogeneity between them. For example, Emery et al. (2003) defined hydraulic habitats using hierarchical clustering algorithm on velocity datasets of the River Tern, Shropshire and River Cole, Birmingham, UK. Hierarchical cluster analysis is most commonly used as an agglomerative process as opposed to a partitioning method. This process begins by treating individual observations as single entities, clusters form and grow based on the similarity between observations until all observations are incorporated into a single cluster. As a result, a hierarchy of clusters are mapped onto a dendrogram, and the optimum number of clusters is defined as the largest spacing between the hierarchy of clusters.

The hierarchical approach is advantageous over other methods such as k-means as the researcher does not need prior knowledge or an indication of the likely number of hydraulic patches within the study environment. The k means algorithm seeds cluster centres (number = k, as defined by the user) randomly through the dataset and observations are assigned to a centroid based on the lowest squared error. Nelson et al. (2014) compared a range of clustering algorithms, including k-means and hierarchical, to visual identification methods when defining patches based on gravel sediment distribution. This study observed that the clustering methods mostly produced more detailed and robust classification of patches than the visual identification methods.

Many clustering algorithms, including the hierarchical clustering and k-means clustering algorithms, are criticised for their imposition of crisp boundaries on the river environment where boundaries are more complex and likely to be represented by gradual transitions. Some studies (e.g. Legleiter and Goodchild, 2005; Wallis et al., 2012; Nelson et al., 2014) have used fuzzy clustering to identify habitat patches, whereby membership to a hydraulic patch becomes probabilistic rather than binary. These studies have also been used to identify transitional zones between distinct patches, Wallis et al. (2012) estimated in a study of a reach on the River Arrow, Warwickshire that between 33-38% of habitat was classified as transitional depending on the flow conditions.

In addition to using fuzzy clustering methods, Wallis et al. (2012) also investigated the use of spatial configuration metrics typically used in landscape ecology to the aquatic environment. These metrics investigated the role of patch characteristics such as shape complexity, size and diversity at a range of flows and observed that these patches were dynamic over space and as flow varied. These more advanced algorithms and metrics may present added detail but also add an additional level of complexity to data analysis.

It is also worth noting that the study by Nelson et al. (2014) was applied to a laboratory environment and the other studies were not applied practically to river restoration monitoring. Consequently, the applicability of these methods for routine river restoration monitoring is unknown.

2.5.3 Physical Habitat Modelling

Physical habitat simulation is a technique that has been used to predict the area of suitable physical habitat of different species based on observed preference criteria (velocity, depth and substrate). Preference criteria are typically scored between 0 and 1 for each variable and averaged to produce a global habitat suitability score (Brown and Pasternack, 2009). An example of this technique is the Physical Habitat Simulation System (PHABSIM) that was developed by the US Fish and Wildlife Service in the 1970s and pioneered by Bullock et al. (1991) in the UK (Spence and Hickley, 2000). PHABSIM uses hydraulic modelling calibrated by measured cross-sections, combined with hydrological modelling to assess the provision of suitable physical habitat area at a range of flows. This method has been used to regulate abstraction and flow releases on many UK rivers to maintain physical habitat provision (Spence and Hickley, 2000). The PHABSIM concept has been further developed to aid the design of river restoration schemes at larger scales (e.g. MesoHABSIM, Parasiewicz (2001)).

Fuzzy approaches have also been applied to physical habitat simulation to account for the uncertainty in species suitability criteria which are often only semi-quantitative and subject to expert judgement (Lane et al., 2006; Mould, 2007; Mouton et al., 2007; Radinger et al., 2017). For example, the Computer Aided Simulation Model for Instream Flow Requirements (CASiMiR) that uses a fuzzy approach was demonstrated by assessing the impact of a weir removal on the River Zwalm, Belgium (Mouton et al., 2007). The simulations suggested that following the removal of the weir, the physical habitat suitability would have improved for all life stages of bullhead. Non-fuzzy approaches have also been used to assess the suitability of river restoration pool-riffle designs for Chinook Salmon on the Trinity River, California, US (Brown and Pasternack, 2008). This approach has been used to assess the post-project provision of physical habitat. For example Koljonen et al. (2013) observed increases in suitable habitat for Atlantic Salmon following the restoration of the River Kiiminkijoki, Finland in 2003. The use of these approaches has been widely used but still typically with hydraulic models rather than real data (e.g.

Koljonen et al., 2013; Tamminga et al., 2015; Holzapfel et al., 2017), possibly due to the lack of spatial and temporal high-resolution velocity data.

These models have been criticised for their inability to evaluate spatial dynamics of physical habitat (Crowder and Diplas, 2002), and their global outputs (such as weighted usable area) may not be sufficient to assess habitat quality (Carnie et al., 2015). For example, Mouton et al. (2007) acknowledged that migratory barriers should be considered when assessing habitat suitability at the larger scale. However, there appear to be limited studies that have considered local scale fragmentation on the quality of physical habitat provision predicted by suitability curves. Some attempts have been made to classify the quality of the physical habitat according to the level of preference by species (i.e. between 0 and 1) as suggested by preference criteria (e.g. Brown and Pasternack, 2009 and Wheaton et al., 2010). However, there may be justification for implementing spatial configuration metrics into physical habitat simulation assessments for river restoration schemes, similar to that demonstrated by Wallis et al. (2012) for hydraulic habitat assessments.

In summary, there are a range of approaches that have been documented within the literature to assess geomorphological and physical habitat performance of river restoration schemes. Some of these techniques have been more widely demonstrated for river restoration monitoring than others. However, the review of the literature suggests that using multiple assessment methods may be beneficial for interpreting geomorphological and physical habitat performance.

2.6 Data Dissemination

The communication of river restoration performance following data collection and analysis is critical for developing best practice for adaptive management and the design of future schemes (Addy et al., 2016). Additionally, river management and restoration could benefit from challenging the perception of non-experts (including the public and not just project stakeholders or scientists from other disciplines) of what rivers should look like and how complex channel forms support rich river ecosystems (Wohl, 2016). Education around why river restoration is taking place and how it has performed may be a help to promote and secure the future of sustainable river management.

Surveys of river restoration practice indicate that in some areas where monitoring does occur, a high proportion of results are not disseminated to stakeholders (O'Donnel and

Galat, 2008, Cokerhill and Anderson, 2014). Reviewers have argued that a lack of clear mechanisms for disseminating river restoration performance may be hindering learning (O'Donnel and Galat, 2008), as practitioners are aware of the benefit of knowledge exchange (Matthews et al., 2010). Skinner and Bruce-Burgess (2005) identified conferences, technical newsletters and peer-reviewed journal articles as potential outlets for widely disseminating results. However, these may not be particularly accessible to all stakeholders. Web data management, databases and performance tracking tools have been identified as potential solutions to this inaccessibility (Neumaan, 2007; O'Donnel and Galat, 2008; Castillo et al., 2016).

This section reviews the different methods currently used to disseminate learning outcomes from river restoration monitoring, these have been classified here into five categories for discussion; i) databases, ii) peer-reviewed journal articles, iii) conferences, iv) guidance literature and v) social media. Databases such as the NRRSS (USA) and NRRI (UK) hold the potential to inform broad scale learning from river restoration performance and a useful resource for practitioners (Chapter 1.3.1). However, these conclusions are sometimes limited if databases are fragmented or data is inconsistent between projects (Vaughan et al., 2009; Castillo et al., 2016). Databases are limited by the information gathered within individual restoration projects and therefore accounting for dissemination must be a consideration during project design (Bernhardt et al., 2005). To improve databases, a simple routine assessment procedure or a standard assessment procedure with a variety of levels dependent on the scale/risk of the scheme would enable the comparison of case-studies. Additionally, access to these databases is not always readily available to academics and practitioners, although in some cases information from the databases may be available on request.

Publications of river restoration monitoring and evaluation in peer-reviewed journals are typically more detailed and lengthy than a database entry. However, publications are not widely available in the public domain to practitioners, unless they are open access. However, the drive for 'impact' in academic research particularly in the UK, as discussed in Chapter 1.4, is increasing the prevalence of open access articles. Even in the case of open access journals, the process of publication, publication fees and embargo periods (the length of which vary dependent on the journal) can delay dissemination and thus the timely incorporation of these outcomes in the adaptive management process. Therefore, in any circumstance, dissemination of river restoration monitoring is likely to incur a financial cost to at least one stakeholder. The scholarly format and specific nature of journal articles can also be a barrier to access, particularly as river restoration is a very interdisciplinary science which engages a wide range of stakeholders.

A major change in research outputs funded by UK research councils occurred in 2015, whereby data used in publications must now be made freely available (Gregory, 2014). Therefore, sharing of data and the public availability of river restoration datasets is likely to improve. However, not all river restoration monitoring undertaken by academics is funded by research councils. This move towards sharing data is encouraging given the challenges faced with 'dark data' (Chapter 1.4). This highlights a need to collect data that is comparable and to share this data in a usable format. There is also a similar need for analytical tools in open source software that are freely accessible to all stakeholders.

The exchange of new knowledge and experiences between a wider range of stakeholders may be facilitated by conferences, depending on the target audience. Yet, conference outputs (e.g. conference proceedings) may have similar access and availability issues as peer-reviewed journal articles. Visual aids, such as PowerPoint presentations, are commonly made available after the conference on the convener's website (e.g. www.therrc.co.uk and www.rrnw.org), or a dedicated conference website (e.g. riversymposium.com). However, the latter case is unlikely to be routinely maintained and would be quickly outdated, or closed. Presentation aids are most often not designed to be interpreted without the guidance of the presenter but may be useful cues to those who have attended the presentation. Therefore, conference outputs are more likely to be used by those who have attended the conference as a visual recall, and not the wider river restoration community. That said, conferences can be a useful tool to gain feedback on monitoring and modes of further analysis that the investigator had not yet considered. They can be a good pre-cursor to publishing a peer-reviewed journal article or non-traditional publication method (i.e. grey literature).

Grey literature guidance is often more readily available and accessible to the wider river restoration community in the UK and beyond. The resources are often produced and maintained by not-for-profit organisations. In the long term, these organisations may lack the financial ability to improve and maintain these guidance resources. Examples of current best-practice in this type of dissemination are two electronic resources maintained by the River Restoration Centre (RRC), namely the Manual of River Restoration Techniques (MRRT) and the EU Riverwiki. The MRRT is an online resource of case-study demonstration projects of specific techniques, whilst the EU Riverwiki is a web-based encyclopaedia of river restoration projects. The latter provides unique features such as the ability for a user to rate case-studies, upload accompanying reports and partake forum discussions which promote stakeholder learning. However, in both resources, the geomorphic and physical habitat analyses of featured schemes are lacking and at present are not fully facilitating the dissemination of river restoration monitoring. This is likely attributed to both a lack of monitoring and the lack of methods or techniques for effectively communicating physical habitat performance (Chapter 1.4).

Social media platforms are also critical methods of communicating river restoration dissemination outcomes that are gaining momentum. Social media supports existing methods of dissemination through promotion. For example, conference attendees regularly take to Twitter to highlight key findings from conference presentations. However,

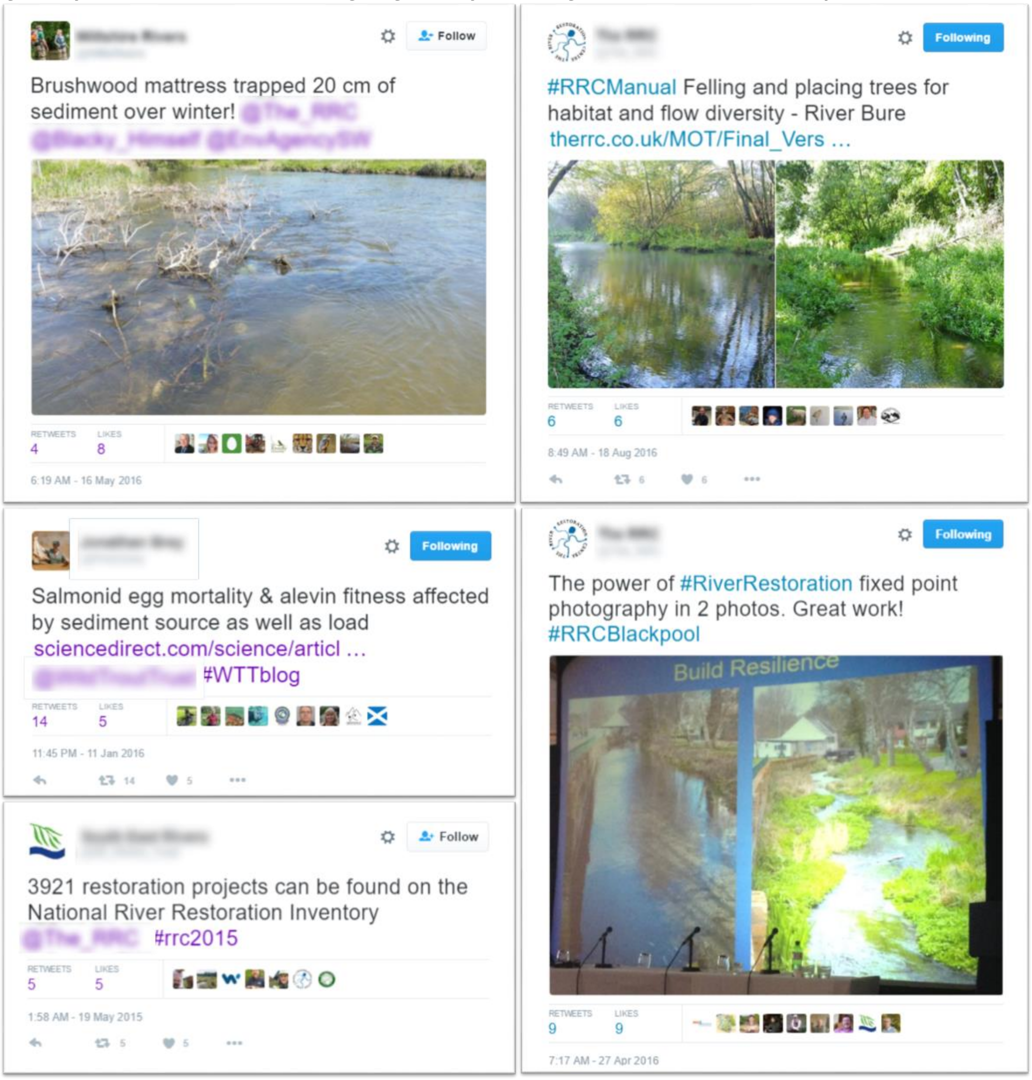


Figure 2.7 Examples of social media postings of learning opportunities for river restoration. From top left clockwise: tweet of learning experience; tweet promoting guidance literature; tweet of learning from conference dissemination; tweet promoting a river restoration database; tweet summarising peer-reviewed journal article.

social media platforms are also an outlet in their own right, that concisely disseminate knowledge to the wider public. Examples of how social media has been used in river restoration are shown in Figure 2.6. Peer-reviewed journals within geomorphology are beginning to acknowledge that social media may have value for communicating research and may be influential on river management practice (e.g. Fox et al., 2016 and Clarke et al., 2017). For example, social media was used as part of a campaign against practitioners by a local community group to prevent the removal of the Bonneville dam on the Swift River, Massachusetts. In another example in river management, social media has been used to regularly update communities and businesses of water levels from gauging station data in the UK (Balch, 2014). There is potentially a niche to develop guidelines to promote the wider use of social media within river restoration dissemination.

2.7 Summary

In summary, this review of river restoration monitoring principles has highlighted that all components of a river restoration monitoring programme are critical in the evaluation process. The collection of adequate baseline data and setting rigorous objectives are essential for assessing river restoration performance. There are a plethora of citizen science and more intensive methods of collecting geomorphological and physical habitat monitoring data. Technology has rapidly advanced over the last few decades and has the potential to efficiently capture geomorphological and physical habitat monitoring data. Photogrammetric methods hold excellent potential for river restoration monitoring as demonstrated by Marteau et al. (2016). However, there appears to be little evidence that these technologies have been applied routinely for river restoration to date and the environments in which they may be applied may be limited in scope.

An alternative method for capturing this data for river restoration monitoring could potentially be the ADCP. This technology has been widely applied for scientific fluvial geomorphology studies but rarely within practical situations like the needs of river restoration monitoring. As with any technology, the ADCP has its limitations and is unlikely to be a complete technological solution. The ADCP has been recommended by Marteau et al. (2016) as an alternative to photogrammetric methods for river restoration monitoring despite any real demonstrable evidence that this would be feasible. This thesis will explore the practical use of the ADCP (and other similar technologies by proxy as they produce similar data) for delivering a data-driven performance assessment of a river restoration scheme by physically undertaking a geomorphological monitoring programme of a case-study scheme.

This chapter has also identified that there are a range of methods available to evaluate geomorphological and physical habitat data. Some studies have published their analytical frameworks as code alongside their research (e.g. Wheaton et al., 2010b), yet generally there is limited guidance for applying the more technical analyses reported within academic studies. There appears to be a reliance on simulating velocity using hydraulic models, and very few studies appear to have simulated physical habitat based on suitability criteria using real data. These physical habitat simulations also appear to be less rigorous when compared to habitat heterogeneity assessments. Carnie et al. (2016) highlighted that assessing the abundance of simulated physical habitat may not be sufficient to assess quality. Wallis et al. (2012) demonstrated the importance of incorporating spatial configuration metrics in the assessment of physical habitat using cluster analyses. However, similar approaches do not seem to have been applied to physical habitat simulations. This concept will be explored later within this thesis through using the case-study monitoring data.

Data dissemination is integral to the river restoration monitoring process. O'Donnel and Garat (2008) argued that monitoring was often undertaken but rarely disseminated due to a lack of mechanisms to facilitate it. This may have been true a decade ago but since then many platforms for disseminating river restoration learning outcomes have emerged. This chapter adds two additional platforms namely databases and social media to those identified by Skinner and Bruce-Burgess (2005). Online databases such as the EU RiverWiki and social media have a significant potential to assist river restoration dissemination to a wide range of technical and non-technical audiences. Arguably, the mechanisms to facilitate the dissemination of river restoration learning outcomes are improving. However, there is a need to develop performance tracking tools to monitor river restoration performance (Castillo et al., 2016). This concept will be explored later in the thesis within Chapter 7.

As previously mentioned in Section 1.5, a case-study will be used here to facilitate the study in meeting its objectives. The case-study used is the River Rother Habitat Enhancement Scheme (RHES) on the River Rother, West Sussex. This chapter has discussed best-practice monitoring techniques and has informed the development of the monitoring programme of this case-study, outlined in Chapter 4. Researchers have stressed the importance of understanding the historical and present catchment conditions of a river restoration scheme so that it may be effectively evaluated (Downs and Kondolf, 2002; Downs et al., 2011; Friberg et al., 2016). Therefore, Chapter 3 outlines the catchment context of the River Rother and the details of the RHES so it may be appropriately evaluated with consideration to wider catchment processes in Chapters 5 and 6.

3 Case-Study: River Rother Habitat Enhancement Scheme, West Sussex

3.1 Introduction

The lack of monitoring data from river restoration schemes constrains the potential for 'learning by doing' in science and practice, which can lead to ineffective or inefficient restoration schemes. Technological advancements have, in theory, improved the efficiency of undertaking rigorous monitoring of river restoration projects, yet projects using these new technologies are seldom reported. This chapter introduces the case-study restoration scheme that was identified from the Catchment Restoration Fund (CRF) in the pursuit of fulfilling the objectives of this study outlined in Chapter 1.5. Defra allocated £24.5 million for river restoration projects under the CRF shortly before the start of this research project in 2012 to support the delivery of EU WFD targets.

The CRF funded 42 restoration schemes across England with guidance from the RRC (Defra, 2013) which were required to complete construction in 2013. Therefore, there was a high level of certainty a CRF scheme would be completed within the short time frame of this research project. The scheme chosen for this research study was a reach scale enhancement scheme on the River Rother, West Sussex that was funded by a grant awarded to the Arun and Rother Rivers Trust. The scheme aimed to restore spawning and rearing habitat for brown trout (*Salmo trutta)* by directly modifying physical habitat. This site was selected for the following reasons;

- Restoration was proposed at the reach scale;
- Direct physical channel modification was proposed in design;
- The proposed design was somewhat experimental;
- Afforded an adequate period of post-project monitoring.

Understanding the catchment context, including historical conditions, is critical in designing and evaluating the performance of a river restoration scheme (Downs and Kondolf, 2002; Downs et al., 2011; Friberg et al., 2016). First, the chapter examines the wider context of the catchment with a view to understanding the rationale for the restoration scheme within its physiographic setting. Second, the chapter presents a justification for high-resolution monitoring of the RHES. The discussion in this chapter builds on a geomorphological assessment of the catchment that was undertaken for the National Rivers Authority (NRA, now the Environment Agency (EA)) by Sear (1996) with 20 years of additional information and new resources. This updated baseline information is reported within this thesis and in more extensively in Cox and Soar (2017), but post-

Table 3.1 Datasets used in the Rother catchment geomorphological assessment

Section	Information	Source(s)
Hydrology	Daily gauged flow data (Iping and Hardham gauging stations)	*National River Flow Archive (Centre for Ecology and Hydrology)*
	Hydrology Assessment	*Jeremey Benn Associates (JBA)*
	Flood Risk Model	*Environment Agency*
Geology	BGS 1:10 000 Geology	*Edina*
	BGS 1:50 000 Geology	*Edina*
	Soilscapes	*South Downs National Park Authority*
Topography	Open Data 2m LiDAR	*Environment Agency*
	Ordnance Survey 5m DTM	*Edina*
Geomorphology	Flood Risk Model	*Environment Agency*
	Rother Navigation Plans & historical maps	*Petworth House Archives*
	Ordnance Survey First Series 1842-1952 (1:10 560)	*Edina*
	Open Data Oblique Photography	*Environment Agency*
	Open Data 1m LiDAR	*Environment Agency*
	Aerial Imagery	*Google Earth*
	Historical photographs	*Mills Archive, Geograph, Britain from above*
Land Cover and Land Use	Rural Payments Agency (RPA) data	*Environment Agency*
	Ancient Woodlands	*Environment Agency*
	Corrine Land Cover	*European Environment Agency*
	Tithe Maps from West Sussex and Hampshire	*West Sussex and Hampshire Country Record Offices National Archives*
	Rother Navigation Plans	*Petworth Archives*
	Historical Maps	*Edina*
	England Urban Areas (2001)	*Edina*
	Land Cover Map (2007)	*Centre for Ecology and Hydrology*
	Population data	*Vision of Britain through time*
Catchment Initiatives	National Grid Resurvey (1:2500)	*Edina*
	Historical Reports	*Environment Agency/Rivers Trust*
	Personal communication with local land managers	
Ecology	Annual fisheries survey	*Environment Agency*
	WFD assessments	*Environment Agency*
	Various ad-hoc surveys (including tree, invertebrate, otter surveys etc.)	*Rivers Trusts/NGOs*

dates the information used to design the RHES. The catchment overview in this chapter was undertaken considering the principles of a fluvial audit (Sear et al., 1995; Sear et al, 2010). These assessments aim to provide a semi-quantitative assessment of the geology, hydrology and geomorphology of a catchment to offer a contextual basis for sediment management problems (Sear, 1995; Sear et al., 2010). Importantly, these assessments provide essential baseline geomorphological information that can be used to identify areas for restoration and inform restoration design (Thorne et al., 1997). The data used to inform the overview of the catchment within this chapter (summarised in Table 3.1) have been processed in ArcMap 10.3.1 to produce thematic maps.

In addition, these data have been used in a stream power assessment to assess broad-scale catchment sediment transport processes (Wallerstein et al., 2006). This method has been previously used to inform restoration along a tributary in the upper catchment (Brookes et al., 2005; Wallerstein et al., 2006). An assessment of stream power at the catchment scale of the River Rother is not thought to have been undertaken to date. Therefore, this chapter provides novel baseline evidence for targeting and designing future restoration activities that may be considered within the catchment. The information reported within this chapter is used later in Chapters 5 and 6 to evaluate the performance of the RHES. It is important to note that a broad aim of the scheme was outlined at the start of the project, specific quantitative performance criteria were not defined to the author's knowledge.

3.2 River Rother, West Sussex - Catchment Overview

The River Rother, West Sussex, is a tributary of the River Arun that drains a catchment of approximately 350km^2 (Fig. 3.1). The River Rother is also referred to as the Western Rother to avoid confusion with the River Rother in East Sussex. The source of the River Rother in West Sussex is a spring near Noar Hill, Hampshire. The river flows south through Liss to Petersfield, then flows eastwards through Midhurst to Hardham where it joins the River Arun as a right bank tributary. The River Arun subsequently enters the English Channel at Littlehampton. From the source to the confluence with the River Arun, the River Rother flows for approximately 56 kilometres. The River Lod is a significant left bank tributary that joins the River Rother in the lower catchment near Selham, where it accounts for approximately 15% of the total drainage area.

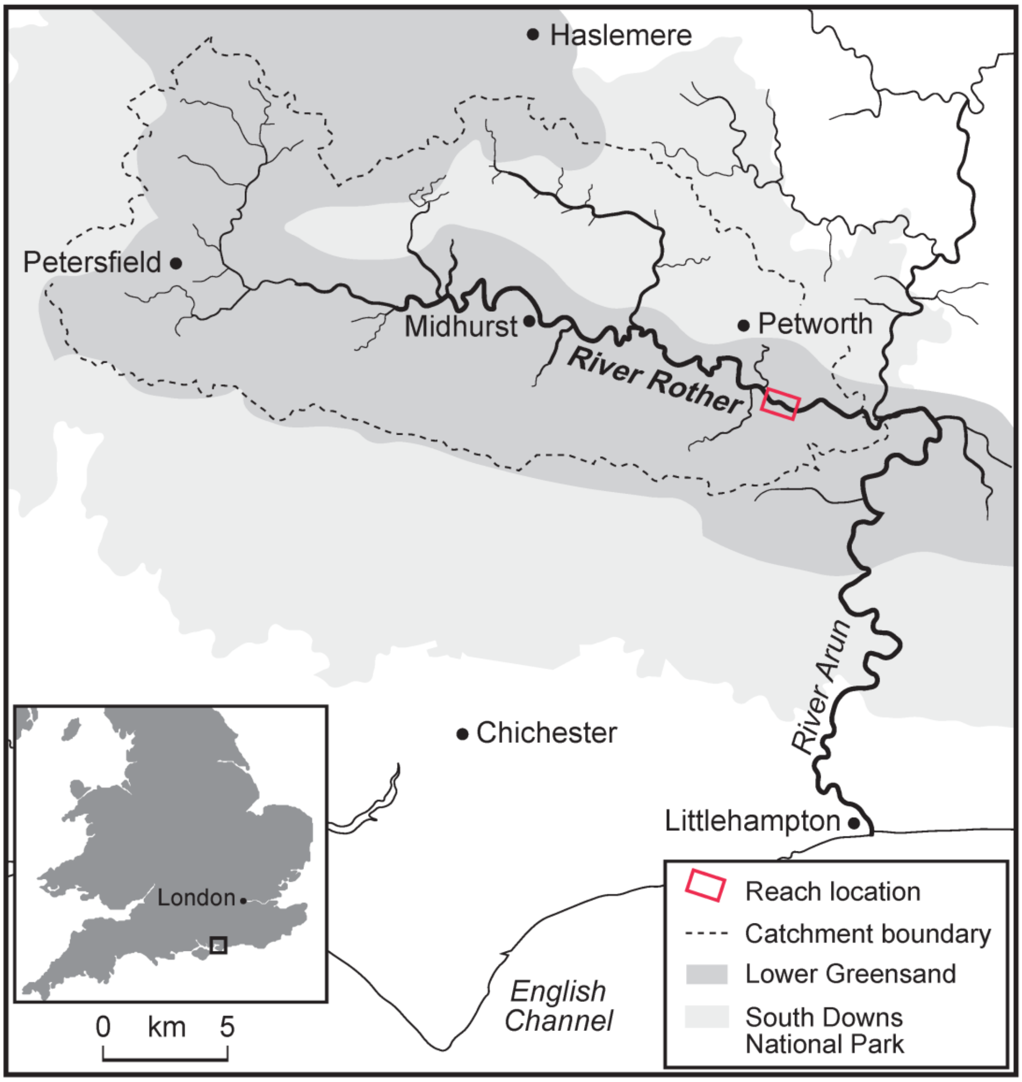


Figure 3.1 Location of River Rother West Sussex, identifying location of the RHES within the catchment and South Downs National Park boundaries. Map drawn by E. Watts (University of Nottingham).

The topography of the catchment (Fig. 3.2) and the wider South Downs has primarily been shaped by the action of fluvial processes (Jones and Robins, 1999). The wider catchment of the River Arun is thought to have once extended further into the English Channel as part of a much larger river system during periods of lower sea level in the Quaternary (Gupta et al., 2004). The present catchment has a minimum elevation of 3.9 m near the confluence with the River Arun, and a maximum elevation of 278 m near Blackdown in the North of the catchment. The Rother Valley is confined by the chalk escarpment of the South Downs to the south and west and the Surrey Hills to the north. As such, the River Rother flows through the heart of the South Downs National Park, an iconic area of chalk downland (Fig. 3.1).

It is the only river which has a catchment that falls completely within the National Park's boundaries. The South Downs National Park is perhaps atypical of National Parks which usually are designated for their natural environment. There is a distinct human influence on the landscape of the South Downs which has significant cultural heritage value. The designation of National Park status to the area was delayed from the original proposition in the early twentieth century until 2009, due to the emergence of extensive and intensive agricultural practices. However, the South Downs National Park Authority (SDNPA) has been fully operational since 2011 and landscape restoration is an important priority in their work (SDNPA, 2013). In the intermediary period between the proposition and designation of National Park status, the importance of the area was recognised as an Area of Outstanding Natural Beauty despite the significant anthropogenic influences.

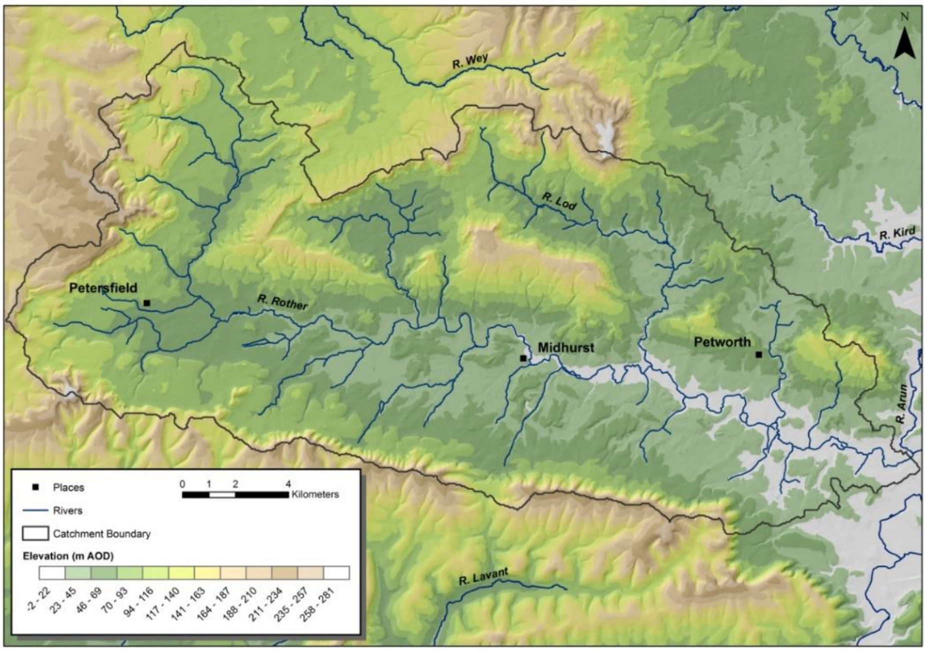


Figure 3.2 Topography of the River Rother Catchment

The River Rother was identified to have a 'poor' ecological status in 2009 according to EU WFD standards (EA, 2017). Invertebrates score highly in WFD assessments but a low density of fish is a concern in the catchment (EA, 2017). Physiochemical indicators of ecological quality such as dissolved oxygen, ammonia and pH indicate a good water quality (EA, 2017). This suggests physical habitat modification may be a primary factor in the decline of the ecological community. Physical habitat degradation has been perceived to have resulted from both the smothering of spawning gravel habitat by fine sediment from agricultural run-off and a lack of habitat heterogeneity due to channel modifications

(EA, 2002; 2013, Boardman et al., 2009). The following sections outline the modifications to catchment processes that may have contributed to the river's poor ecological status.

3.2.1 Geology and Soils

The catchment has a mixed geology which has been largely influenced by changes in sea level since the Cretaceous period. The chalk dome of the South Downs landscape was formed from calcareous deposits in a warm shallow sea environment during the late Cretaceous period, 65 million years ago. The Rother Valley was formed through the erosion of this feature, but the chalk geology remains in the form of an escarpment along the Southern and Western extent of the catchment. In the Rother valley, erosion of the chalk dome exposed older bedrock formed in the early Cretaceous period, 145 million years ago (Hopson et al., 2008), which primarily consist of sandstones and mudstones belonging to the Wealden, Selbourne, Lower Greensand groups (Fig. 3.3 & 3.4). Approximately half of the catchment is dominated by Lower Greensand Group which was also formed in a shallow marine environment (Shand et al., 2003).

As the main river channel follows over the Lower Greensand Group, which comprises primarily sandstones, soils in the local vicinity of the main river are largely sandy and at a high risk of erosion (Fig. 3.5). However, major tributaries such as the River Lod and Hammer Stream flow over the Wealden mudstones which are associated with soils containing a larger clay fraction. Evans et al. (2017) estimated that over 60% of the soils within the wider catchment are at either a high or moderate risk of erosion, the highest risk of those being the Frilford, Fryfield 1 and Fryfield 2 associations. Current land use practices are thought to be contributing to substantial soil erosion within the catchment (Boardman et al., 2009; Boardman, 2013; 2015; 2016).

A study of soils in the Woolbeding area suggests that there may have been two distinct phases of soil erosion within the River Rother catchment; an older period of erosion associated with forest clearance and a period of erosion associated with modern agricultural practices (Wood and Farres, n.d.). The geology and soil information suggests that fine sediment is a natural sediment input to the system. Gravels are found within the system, but in a scenario where fine sediment inputs to the system were to be reduced, it is not clear that a sufficient quantity of gravel could be delivered from the upper catchment to sustain a stable gravel-bed morphology in the lower catchment.

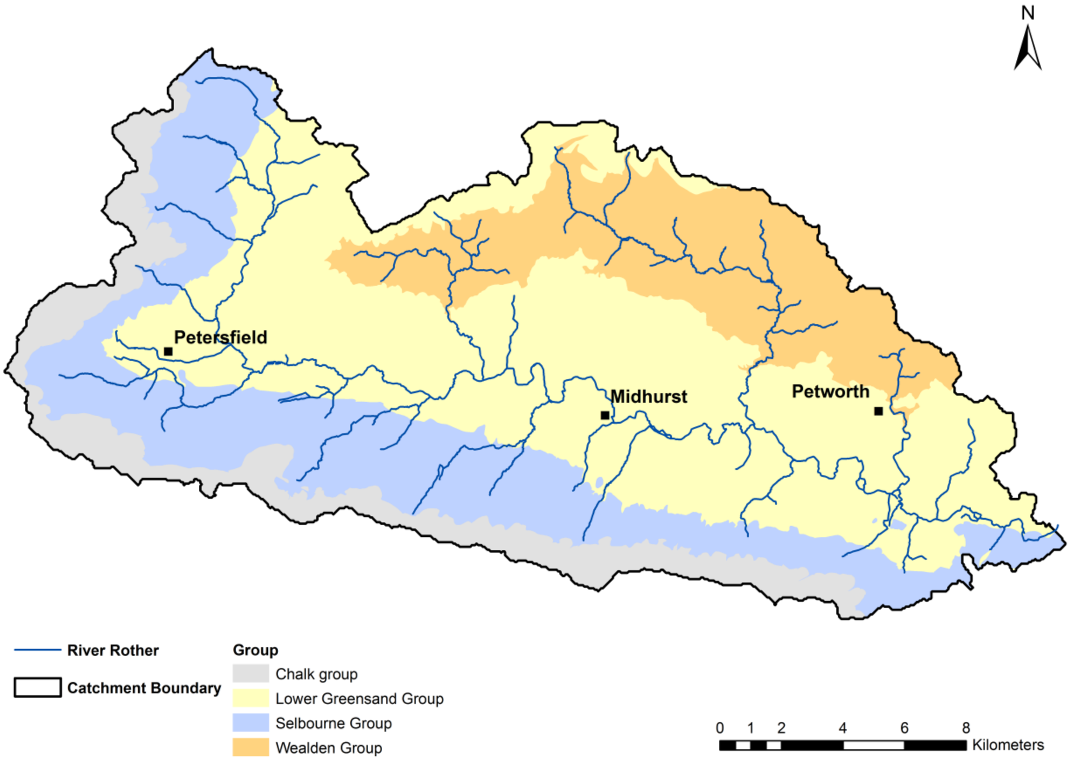


Figure 3.3 Geological Groups in the River Rother Catchment (Data source: Edina Digimap).

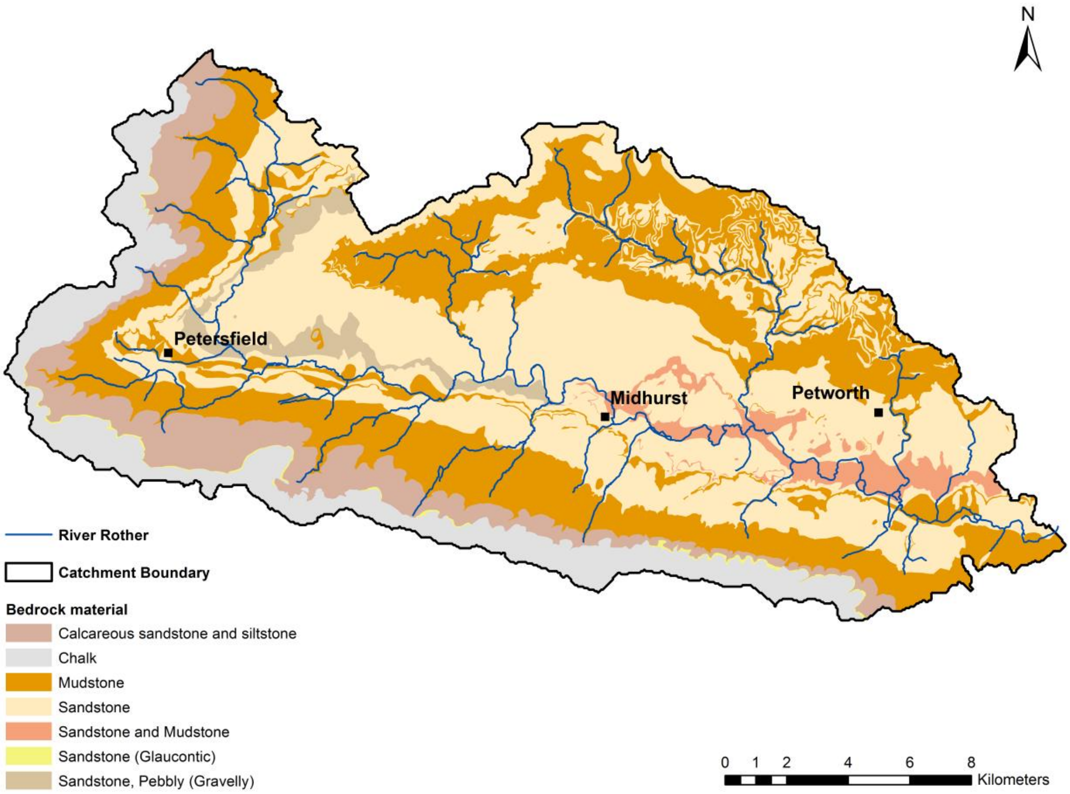


Figure 3.4 Bedrock material in the River Rother Catchment (Data source: Edina Digimap).

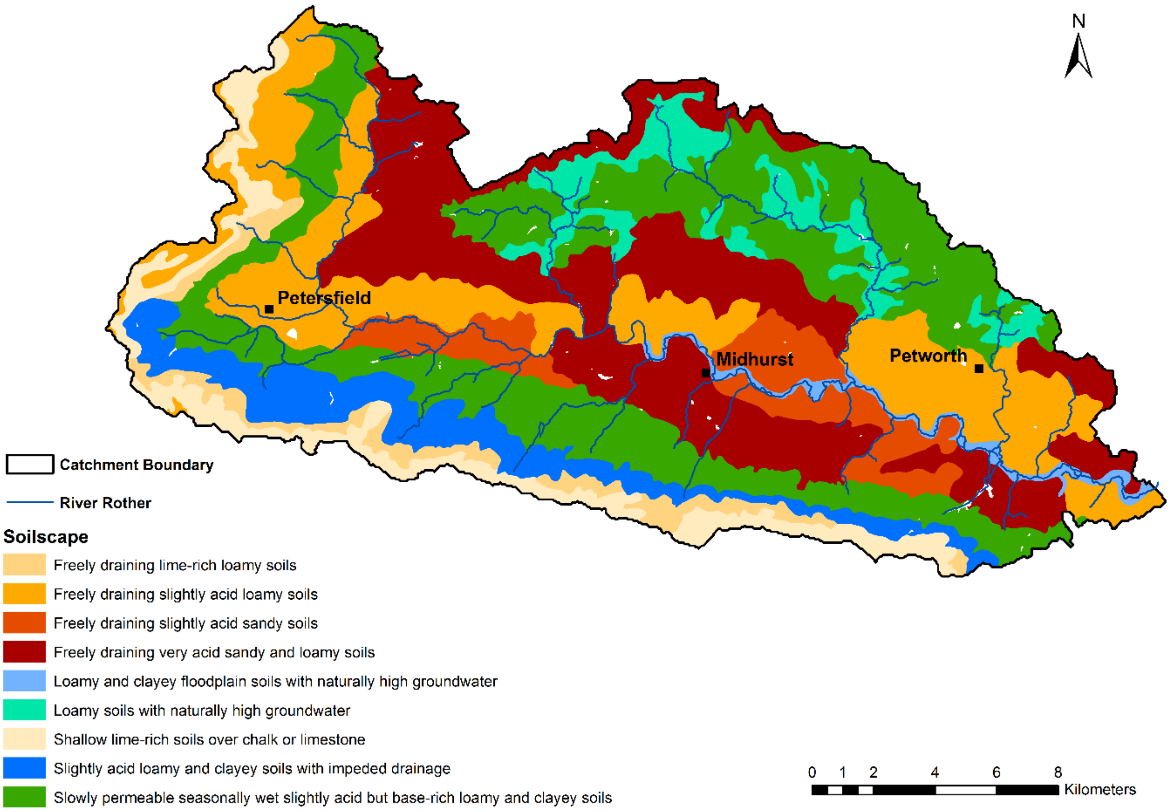


Figure 3.5 Soilscapes of the River Rother Catchment (Data Source: South Downs National Park Authority)

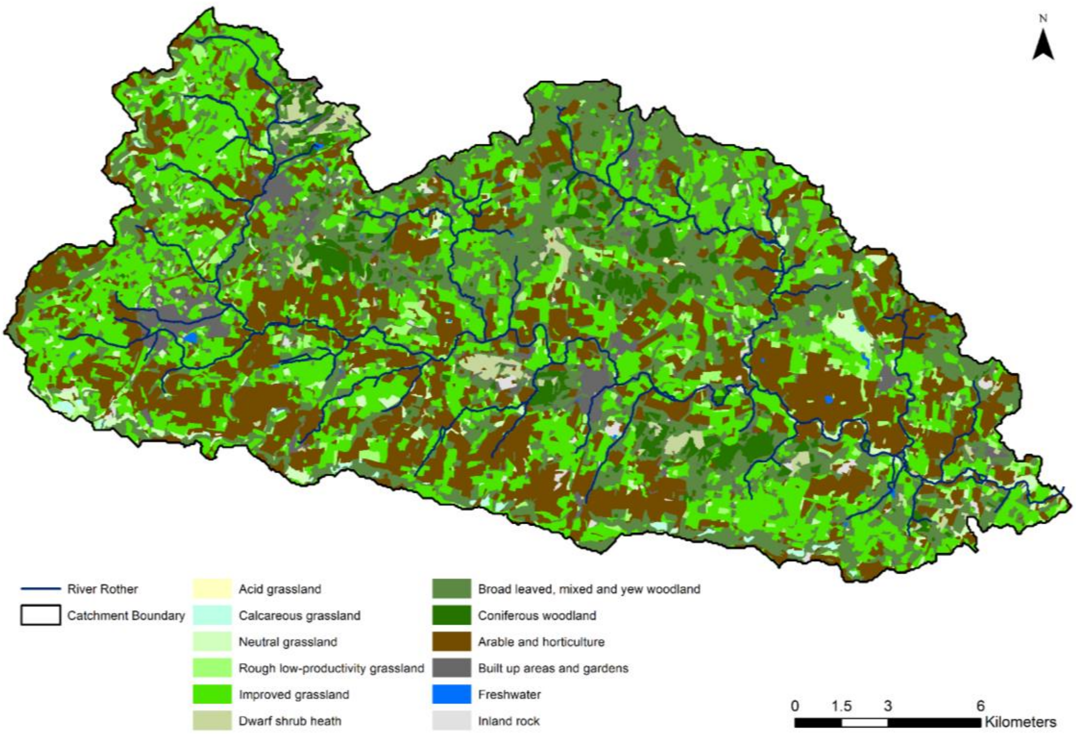


Figure 3.6 Land use of the River Rother Catchment dominated by improved grassland. Data source: LCM2007 © NERC (CEH) 2011. © Crown Copyright 2007. Ordnance Survey Licence

3.2.2 Hydrology

The River Rother is primarily a groundwater-fed waterbody from the greensand and chalk aquifers (PBA, 2007). In combination with these aquifers, sewage treatment works along the river maintain a constant year-round flow of water. However, the flood hydrograph is characteristically flashy due to the impermeable clay soils in the north of the catchment and land uses that are likely to have increased the rate of run-off (see section 3.2.1). The catchment has an average annual rainfall estimated as ~ 900 mm based on data from 1961-1990 (CEH, 2016), with a rainfall gauge at Petworth Park estimating 857mm year (Boardman et al., 2009). A recent analysis of rainfall data within South-East England, which includes a dataset from Petworth, suggests that climate change is contributing to a trend of less frequent but more intense periods of rainfall (Burt et al., 2016).

The river is gauged at five locations within the catchment (in downstream order), Princes Marsh (since 1972), Iping Mill (since 1966), Cocking (Costers Brook, since 1973), Lodsworth (River Lod, since 1970), Fittleworth (since 1981) and Hardham (since 1959) (PBA, 2005). Discharge is gauged at Hardham using a compound crump weir, installed in 1982 at 3.9m AOD, to replace a pre-existing weir (CEH, 2016). The present gauging station measures discharge up to 16 $m^3 s^{-1}$; discharges above this value over top the gauging station and therefore cannot be measured with any accuracy. Spot measurements are undertaken at discharges above 16 $m^3 s^{-1}$ approximately 3km upstream at Fittleworth to supplement the Hardham gauging station dataset.

The flow duration curve for Hardham Gauging Station (Fig. 3.7) is based on data from between 1959 and 2014, the curve indicates a seasonal variation in flow with a median flow duration of approximately 2 $m^3 s^{-1}$ during summer and 5 $m^3 s^{-1}$ during winter. The flatness of the curve at lower flows is attributed to the influence of the groundwater aquifer and sewage treatment effluence on sustaining flow year-round (Cox and Soar, 2017). However, the steeper gradient of the curve at higher flows reflects the flashiness of the flow regime. The flow duration curve for the gauging station at Iping Mill has a more constant gradient indicating groundwater is a primary control of the hydrograph in the upper catchment (Fig. 2.7). On the other hand, higher flows in the lower catchment are likely driven by flashy flows from the River Lod catchment that is dominated by a clay geology (PBA, 2007).

The River Lod catchment is inherently more responsive due to the underlying impermeable clay geology. However, assessment of flow duration data by Howarth and

Manning (2001) has suggested that the River Lod catchment has become more responsive because of an increase in arable land cover since the 1970s. This land use trend (discussed further in Section 3.2.3) has also been observed more widely within the catchment (Sear, 1996), which may suggest that wider River Rother catchment has also become more responsive over the last 50 years. However, the human influence on the catchment's hydrology may extend much further, as historical evidence suggests that the river has been regulated by mills and ponds since the 11th century. For example, whilst the Rother Navigation was active during the early 19th century (see Section 3.2.4), mills were used to maintain a minimum water depth of ~ 1m in the main channel (Vine, 1995).

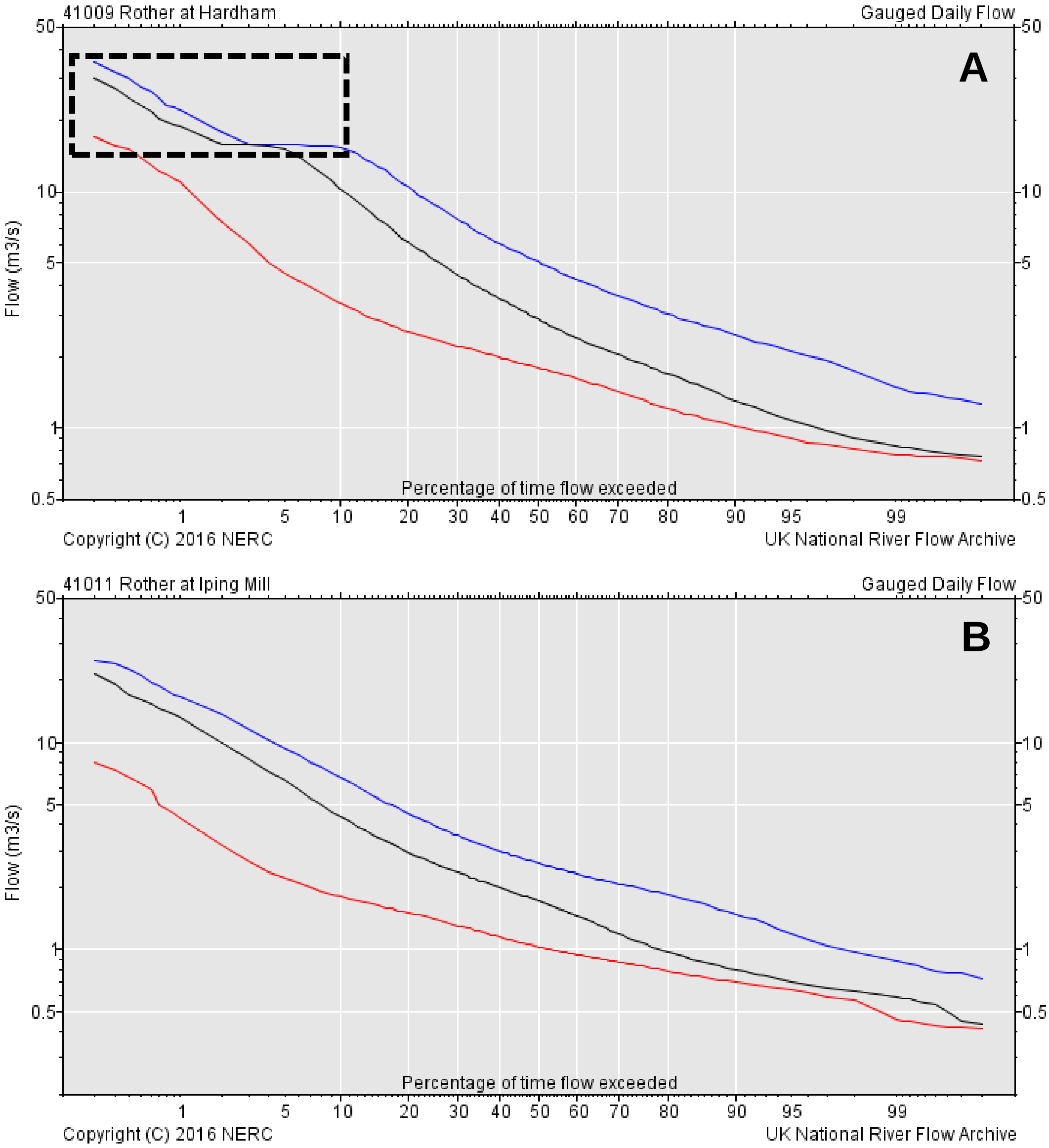


Figure 3.7 Flow duration curve for the Hardham (A) and Iping Mill (B) gauging stations from the National River Flow Archive (CEH, 2016), Red = summer flows, Black = annual average flows, Blue = Winter flows. The dashed box highlights the flow at which Hardham weir is overtopped and the uncertainty around these estimated flows significantly increases.

Some significant high flow events have been recorded within the catchment since the mid-20th century, which may have caused significant geomorphological change and resulted in episodic sediment inputs. High flow events recorded in 1960, 1968 and 1974 caused widespread flooding within the catchment, with the Petersfield, Stedham, Midhurst, Lodsworth, Petworth, Fittleworth areas notably affected (PBA, 2007). Flows exceeding 70 $m^3 s^{-1}$ were recorded at Hardham Gauging Station (prior to the re-configuration of the weir) during the 1968 flood event (CEH, 2016), where the 1% flow has been estimated at 40.66 m^3s^{-1} (Cox and Soar, 2017).

Over the last two decades, several significant flood events have impacted upon the area, which may have caused episodic increases of sediment inputs into the system. These include the 2013/14 winter floods which resulted in multiple out of bank flows between December 2013 and March 2014 (Fig. 3.8). This event was caused by heavy rainfall from a series of low-pressure systems over the UK (Thorne, 2014; EA, 2016). During the floods, a maximum flow of approximately 47 $m^3 s^{-1}$ was recorded at Iping Mill gauging station. This is a significant flow considering the 1% exceedance flow is estimated at 18.68 m^3s^{-1} (Cox and Soar, 2017). The duration of the flood event at the Hardham gauging station can also be seen in Figure 3.8 despite the capped flows due to overtopping.

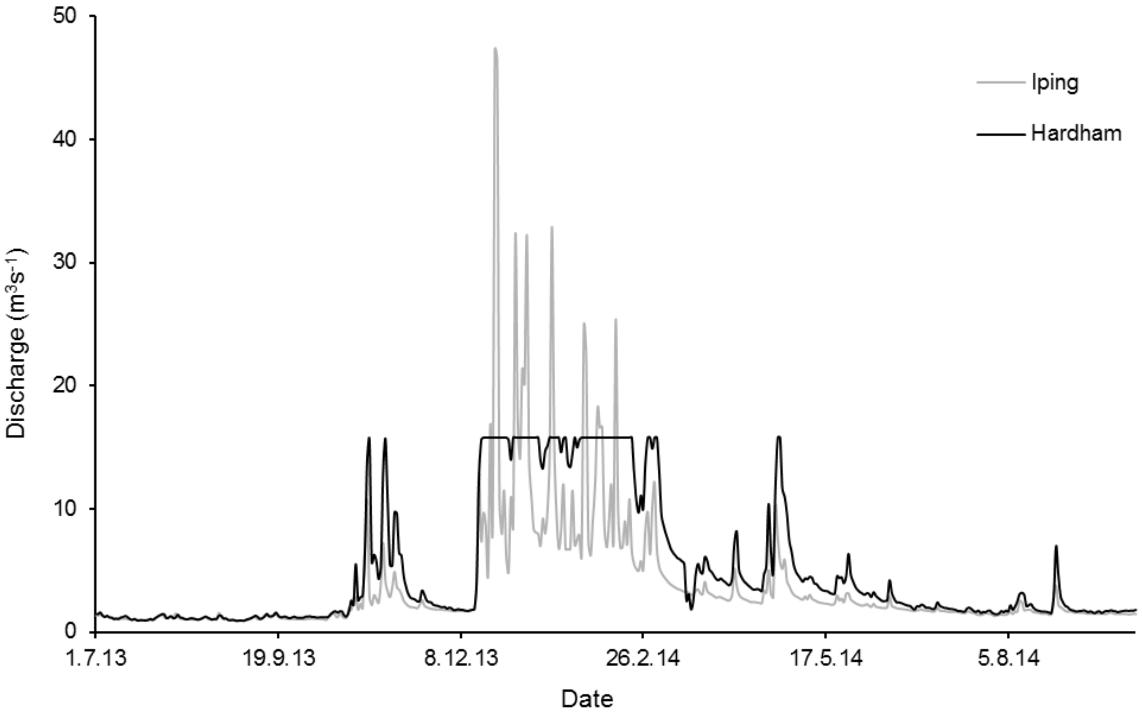


Figure 3.8 Daily flow data from the NRFA between 1st July 2013 and 30th September 2014 for the River Rother at the Hardham and Iping Mill gauging stations

The floods widely impacted on the area and, as discussed later in the thesis, on the research undertaken as part of this study. In England, these floods had a widespread impact with a total economic cost estimated at £1.3 billion (EA, 2016). The local economic impact of the 2013/14 floods in the River Rother catchment is unknown. However, the event resulted in the flooding of numerous properties, soil erosion on agricultural land in the Easebourne and Midhurst areas (CH2M HILL, 2015) and damage to areas of cultural heritage in the Petworth area (Coutlershaw Trust, 2014). There is generally a low flood risk with the River Rother catchment in comparison to other catchments within the UK as the land is primarily agricultural and there are only small pockets of urban areas. However, there is a significant flood risk to properties within the wider River Arun catchment, most notably near Arundel and Littlehampton (EA, 2009). This should be considered when undertaking future restoration within the River Rother catchment.

3.2.3 Land Use

Land use within the catchment is an influential factor on the geomorphology and ecological status of the River Rother (Sear, 1996). Human occupation has been a long-standing characteristic of the catchment, dating back at least 10 000 years (Harris, 2010). Consequently, the land within the catchment has been utilised for a range of industries including agriculture, sand mining and iron works. Whilst the impact of human occupation is evident in the landscape, urban areas only occupy a relatively small 5% of the catchment area. However, these areas have rapidly expanded since the early 20th Century and notably over the last 60 years as areas within the catchment have become popular commuter towns for London (Cox and Soar, 2017). Research suggests that changes in hydrology resulting from urbanisation may have contributed to geomorphological change within the catchment (Brookes et al., 2005).

Agriculture is the dominant land use within the catchment, organised agricultural land use to optimise productivity can be traced back to at least the tenth century (Gardiner, 2000). Pasture accounts for ~ 37 % of the total land cover and is typically found on the upper slopes in the catchment, except for a corridor along the main channel of the river (Fig. 2.9). It is possible that this land use may be a source of run-off that contributes to soil erosion lower in the valley (Farres and Wood, 1990; Wood and Farres, n.d.). Grazing can result in soil compaction which has the potential to decrease infiltration and increase run-off and erosion rates (Shah et al., 2017).

Conversely, the corridor of pasture along the main channel of the river may act as a buffer for catchment run-off. It has been suggested that this area of pasture has been a permanent feature of the floodplain over the last century (Sear 1996). Historical maps suggest that this area of land adjacent to the river has been previously largely occupied by watermeadows for several centuries (Pearson et al., in prep). The decline of watermeadows may have influenced floodplain sediment retention and bank stability (Micheli and Kirchner, 2002a; 2002b; Cutting et al. 2003; Cook et al., 2015). Consequently, it is conceivable that agricultural floodplain land use change may have also influenced wider catchment processes.

In the wider catchment, a decline in pasture and a rise of arable land-use has been observed since the 1970s (Sear, 1996; Howarth and Manning, 2001). At present, arable agriculture is estimated to cover ~27% of the catchment. This land use is widespread throughout that catchment but largely found in closer proximity to the river than the main bulk of pasture (Fig. 3.6). It is both the increase and changes to arable agricultural practices since the 1970s that have been recently considered a significant source of fine sediment generation within the catchment (Sear, 1996, Boardman et al., 2009). The crops cultivated within the catchment (e.g. potatoes) are particularly problematic as they expose bare soil and give rise to erosion processes such as soil crusting (Farres and Wood, 1990; Boardman et al., 2009). Soil crusting, which is particularly prevalent in lower greensand soils, may also decrease infiltration rates and increase the potential for run-off and the risk of soil erosion (Farres and Wood, 1990).

Run-off from arable fields susceptible to erosion has been observed to discharge into the River Rother via landscape pathways such as sunken lanes (Farres and Wood, 1990; Wood and Farres, n.d.; Boardman, 2013). Field consolidation has been noted within the catchment over the last 150 years which may have increased sediment and hydrological connectivity (Cox and Soar, 2017). Few arable fields are directly adjacent to the River Rother as the immediate floodplain is mostly used as pasture (Fig. 3.6). Nonetheless, connective pathways through the landscape can act as conduits for transporting eroded material from the fields to the river during high rainfall events, for example, sunken lanes (Boardman, 2013). However, significant dis-connectivity to non-tributary sediment pathways through the catchment has been noted and sediment inputs from these pathways are probably highly episodic (Cox and Soar, 2017). The relative contribution of soil erosion to more continual sources such as bed and bank erosion to the catchment sediment yield has not been established.

Woodland also accounts for a significant 23% of the catchment area. It is highly likely that the catchment has been more extensively covered by woodland in the past, but it has since been cleared for other forms of agriculture (Wood and Farres, n.d.; Bannister, 2010). This is likely to have resulted in erosion of the landscape and increased the responsiveness of the flood hydrograph. Coppicing has been recorded within the catchment since the 18th century (Young, 1808; Webster, 2010). Woodland for plantation has seen an increase in areas of the catchment since the 1990s, yet this is thought to be intensively managed (Howarth and Manning, 2001). Consequently, it has been suggested that this may be a source of sediment and may also be contributing to an increasingly flashy hydrology (Howarth and Manning, 2001). An incident was recorded where significant volumes of fine sediment were generated following logging activity and entered the stream network resulting in a significant number fish deaths at a farm (BBC, 2013). Generally, woodland is an excellent habitat and land use for reducing flood risk, however, these reports would suggest that logging is an industry within the catchment that must be carefully managed.

3.2.4 Channel Geomorphology and Change

The River Rother is a sandy river with patches of gravel, it generally exhibits a sinuous planform. Some reaches in the mid and upper catchment near Petersfield are more meandering with some pool-riffle features that are typical of gravel bed rivers. The gravel in these reaches is likely to be derived locally from bedrock and superficial deposits, but sand is still the dominant substrate. The lower reaches of the catchment, particularly downstream of Midhurst, exhibit only a few gravel patches which appear to be associated with restoration activity. The average bankfull channel width varies between ~ 7-8 m near Liss and increases downstream to ~ 18-20 m at Hardham. The channel slope expectantly decreases downstream from 0.0024 in the upper catchment to 0.0006 near Fittleworth in the lower catchment. Significant fine sediment deposition has been experienced in the lower reaches, particularly upstream of Hardham weir. This area has been dredged historically by the EA to maintain the operations of Southern Water (Southern Water, 2014). However, this accumulation (as observed in other areas in the lower catchment) may have been promoted by an overlarge channel geometry resulting from channel modifications (Cox and Soar, 2017).

A stream power assessment was performed for the River Rother catchment to assess the impact of channel geometry on broad-scale sediment transport processes. Stream power is a geomorphological assessment tool that can be used to predict erosion and deposition processes within a channel. It is a measure of the rate of work that is done by the river to overcome in-channel friction and transport sediment (Wallerstein et al, 2006; Sear et al., 2010). Total stream power (Ω) is the energy expenditure per unit length of the channel and is measured in Watts per metre (W m^{-1}), it is calculated using the equation below where ρ is the density of water, g is acceleration due to gravity, Q is bankfull discharge, S is slope. Specific stream power (ω) is the energy expenditure per unit length of the channel relative to the channel width. It is calculated using the equation below where b is bankful width and measured in W m^{-2}. Based on a study of Western European rivers, specific stream power values less than 15-25 W m^{-2} are likely to be indicative of an aggrading environment whereas values greater than 25-35 W m^{-2} are likely to be indicative of an erosional environment (Brookes, 1987a; 1987b; Wallerstein et al, 2006).

$$\Omega = \rho g Q S \qquad \omega = \frac{\rho g Q S}{b}$$

Total and specific stream power were calculated for a series of reaches along the main channel. Stream Power was not calculated for a reach located between Fittleworth and the confluence with the River Arun as channel cross-sections were not available in this location. The average discharge for each reach was estimated from updated hydrology produced by Jeremy Benn Associates. The average slope was estimated from cross-sections of the 2007 flood risk model provided by the Environment Agency (Fig. 3.9). The cross-sections for this model were extracted to HEC-RAS in which a 1 in 2 year flow (approximate to a bankfull flow) was modelled, the average bankfull width was estimated from this model. HEC-RAS was used in this research study, yet it is acknowledged that HEC-HMS (hydraulic modelling system) is available for more complex modelling of catchment systems.

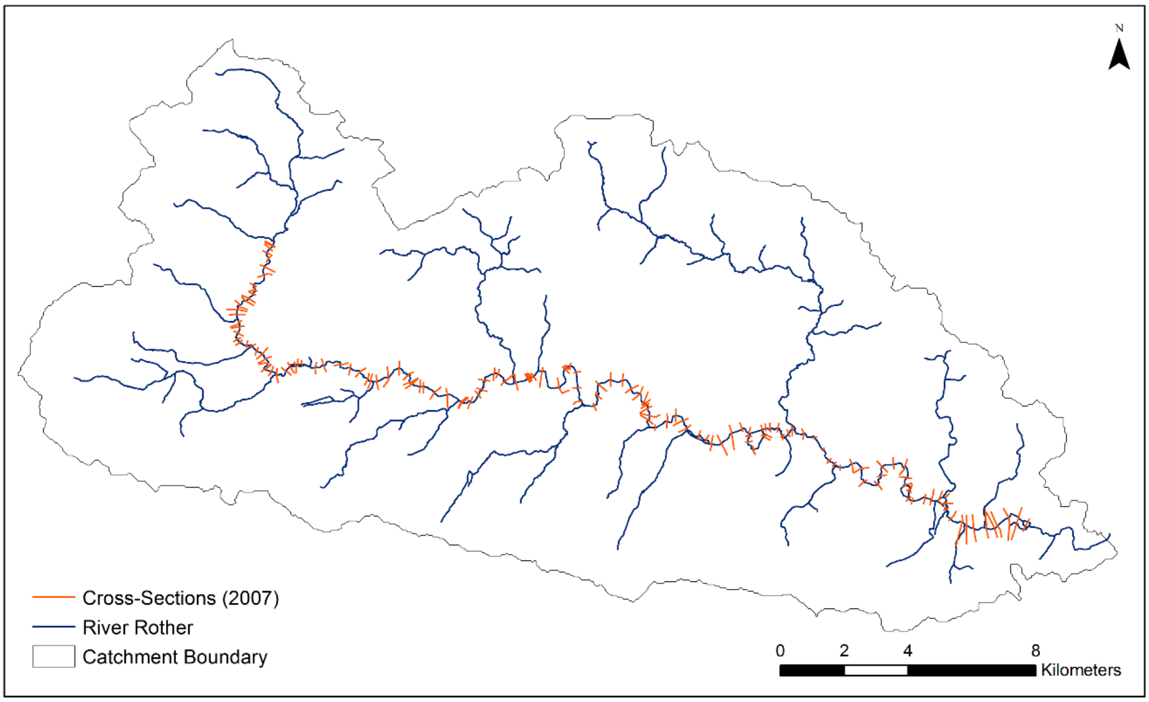


Figure 3.9 Cross-sections from the 2007 flood risk model which were used to populate the HEC-RAS model.

Table 3.2 Total and Specific Stream Power values estimated for the River Rother compared to hydraulic geometry parameters.

	Total Stream Power (W m^{-1})	Specific Stream Power (W m^{-2})	Average Slope	Average bankfull discharge (m^3 s^{-1})	Average bankfull width (m)
Reach 1	208.8	28.2	0.00236	9.0	7.4
Reach 2	256.5	26.2	0.00171	15.3	9.8
Reach 3	371.2	31.3	0.00177	21.4	11.9
Reach 4	259.4	14.4	0.00105	25.3	18.0
Reach 5	489.8	27.6	0.00166	30.2	17.7
Reach 6	349.9	20	0.00103	34.7	17.5
Reach 7	367.1	23.9	0.00102	36.8	15.4
Reach 8	339.8	21.8	0.0009	38.6	15.6
Reach 9	202.1	11.2	0.0004	51.7	18.3
Reach 10	321.5	18.2	0.0006	54.8	17.6

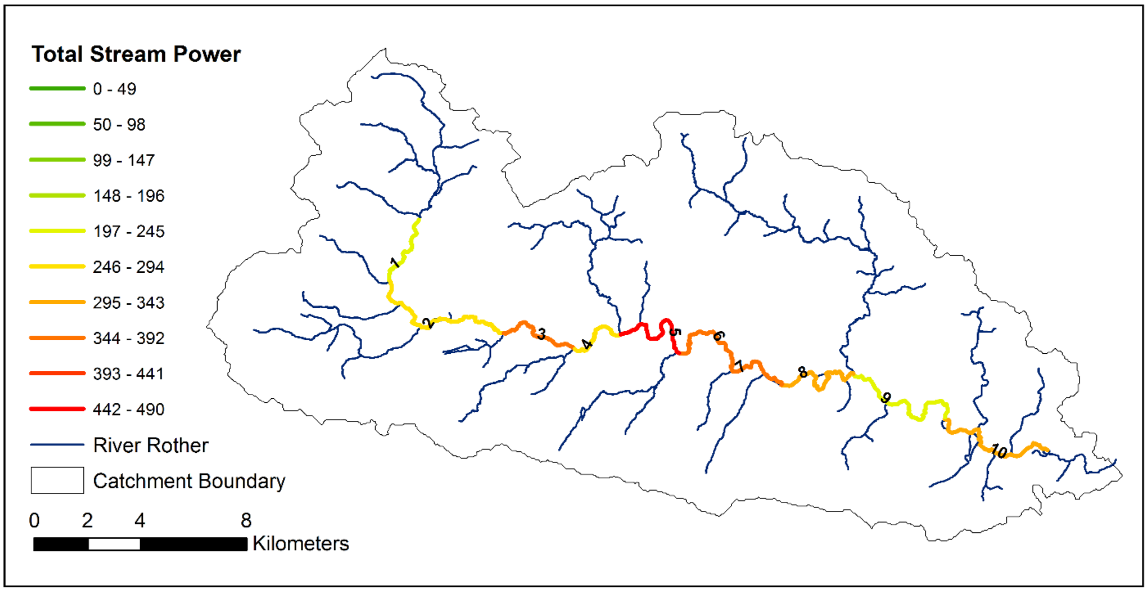


Figure 3.11 Total Stream Power estimates of the River Rother.

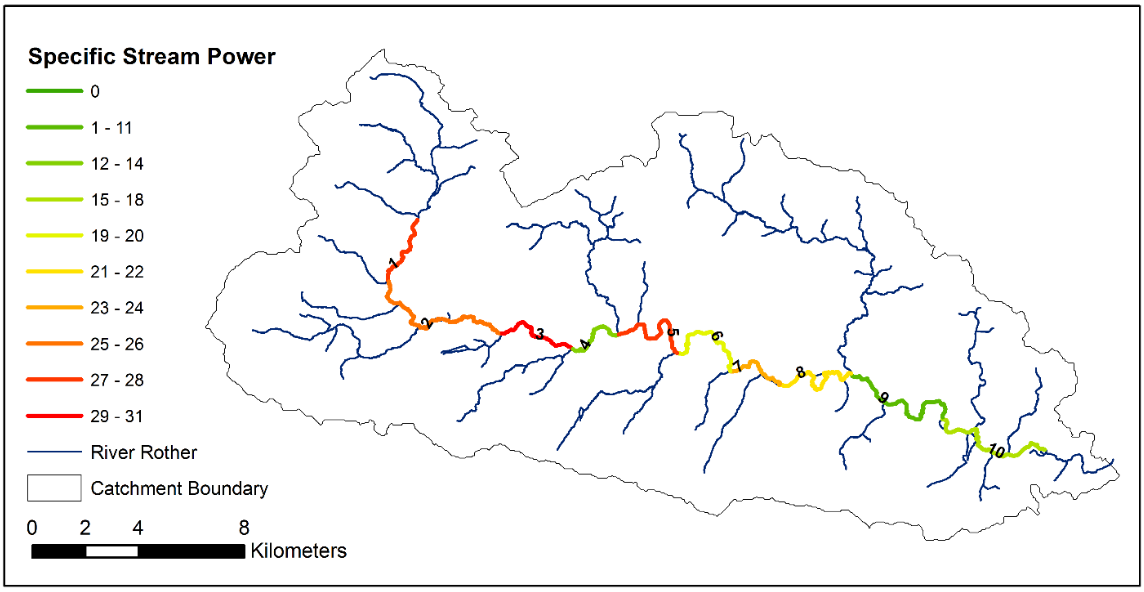


Figure 3.10 Specific Stream Power estimates of the River Rother.

The Stream Power results suggest that channel width may be a primary control on sediment transport processes within the catchment (Table 3.2; Figs. 3.10 & 3.11). In the first two reaches in the upper catchment, the Ω values were discernibly lower than in the reaches located in the mid- and lower catchment (Fig. 3.10, Table 3.2). However, the ω values for the first two reaches in the upper catchment were some of the larger ω values observed within the whole catchment (Fig. 3.11, Table 3.2). This suggests that a narrower cross-sectional geometry may have promoted sediment transport in these reaches, hence the observation of coarser sediment bedforms such as riffles. The slope significantly decreased downstream of Midhurst (Reaches 6, 7, 8, 9 and 10, the RHES was undertaken within Reach 10). However, the increase in discharge appeared to maintain the Ω values observed within the upper reaches. Conversely, the ω values in these reaches were notably lower than those observed within the upper catchment suggesting an overwide channel geometry through these reaches.

Historical evidence suggests the channel has been extensively modified throughout the catchment, but more extensively in the lower catchment where specific stream power is significantly lower. In-channel structures along the main channel of the river have been documented since the mid-11th Century as a mill was recorded near Petworth in the doomsday book (Mills Archive, 2017). Many mills and ponds have since been recorded along both the River Rother and its tributaries (Stidder and Smith, 2001). Many of these features remain as a legacy of a more industrial period and can be identified in the long profile of the river (Fig. 3.12). The long profile (2007) also highlights the decrease in slope from Liss to Fittleworth and a series of steps along the channel which appear to coincide with significant in-channel structures.

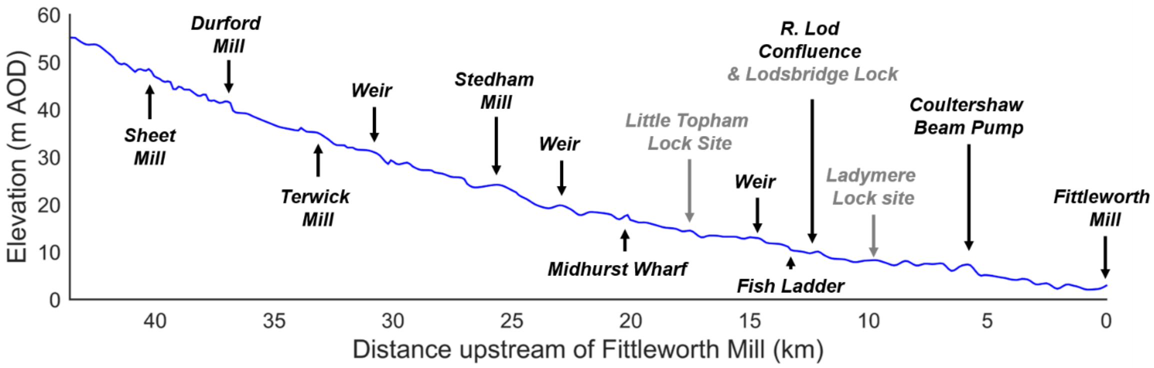


Figure 3.12 Long profile (2007) of the River Rother between Liss and Fittleworth, including significant structures both remaining (black) and removed (grey).

These steps have potentially formed as the result of longitudinal dis-connectivity in sediment transport processes, with sediment accumulating upstream of structures and removed through scour processes downstream. Therefore, it is conceivable that the degradation of the bed in both the main channel and tributaries (resulting from the presence of these structures) may have contributed to increased historic and contemporary sediment loads within the catchment. The number of in-channel structures and other channel modifications may have increased during the 18th Century through the construction of the Rother Navigation, commissioned by the 3rd Earl of Egremont.

It is plausible that in-channel structures that may have restricted passage to barges (e.g. weirs) were removed to enable navigation (Vine, 1995). Construction of the navigation began in 1791, 7 locks were constructed along the river that cut through several meanders, this reduced the channel length by 2 miles between Midhurst and Hardham (Fig. 3.13; Vine, 1995). The navigation plans indicate that the channel of the River Rother downstream of Midhurst was significantly widened to accommodate barge traffic. Fine sediment issues are evidenced in the lower catchment in the decades after the construction of the navigation and well into the early 20th century (Cox and Soar, 2017). This suggests that in-channel fine sediment issues were a concern prior to the recent intensification of agriculture. In addition, prior to the Rother Navigation, historical records indicate the river occupied a narrow channel with "shoals" (Vine, 1995). This indicates the potential presence of large sand deposits in the channel prior to significant channel modifications and the intensification of agriculture in the twentieth century. This implies that sand accumulation may have been a constant feature of the River Rother for several centuries.

In addition to channel widening during the navigation period, the banks were raised above the floodplain which likely reduced lateral connectivity within the system. The banks were raised so significantly that this may have promoted in-channel sediment storage rather than floodplain storage (Cox and Soar, 2017). Floodplain cores demonstrate significant historical deposits (He and Walling, 1996; Walling and He, 1998) and that the rate of sediment deposition on the floodplain potentially decreased between 1966 and 1999 (Walling, 1999). Oblique aerial imagery suggests that the raised bank morphology did not significantly change within this period. However, an analysis of channel change indicates that lateral bank erosion may have also increased between 1975 and 2011 (Cox and Soar, 2017). Conversely, between 1870 and 1975, lateral channel change indicated that in-channel deposition was a dominant process throughout the catchment. These observations may be indicative of several catchment processes, including:

- A possible increase in sediment supply during the earlier time period from infrastructure development projects such as railways.
- A recent increase in the flashiness of the flood hydrograph through land use changes.
- A decrease in bank stability through invasive species (e.g. Himalayan Balsam and signal crayfish) and changes in land use practices (e.g. watermeadows to unfenced grazing).

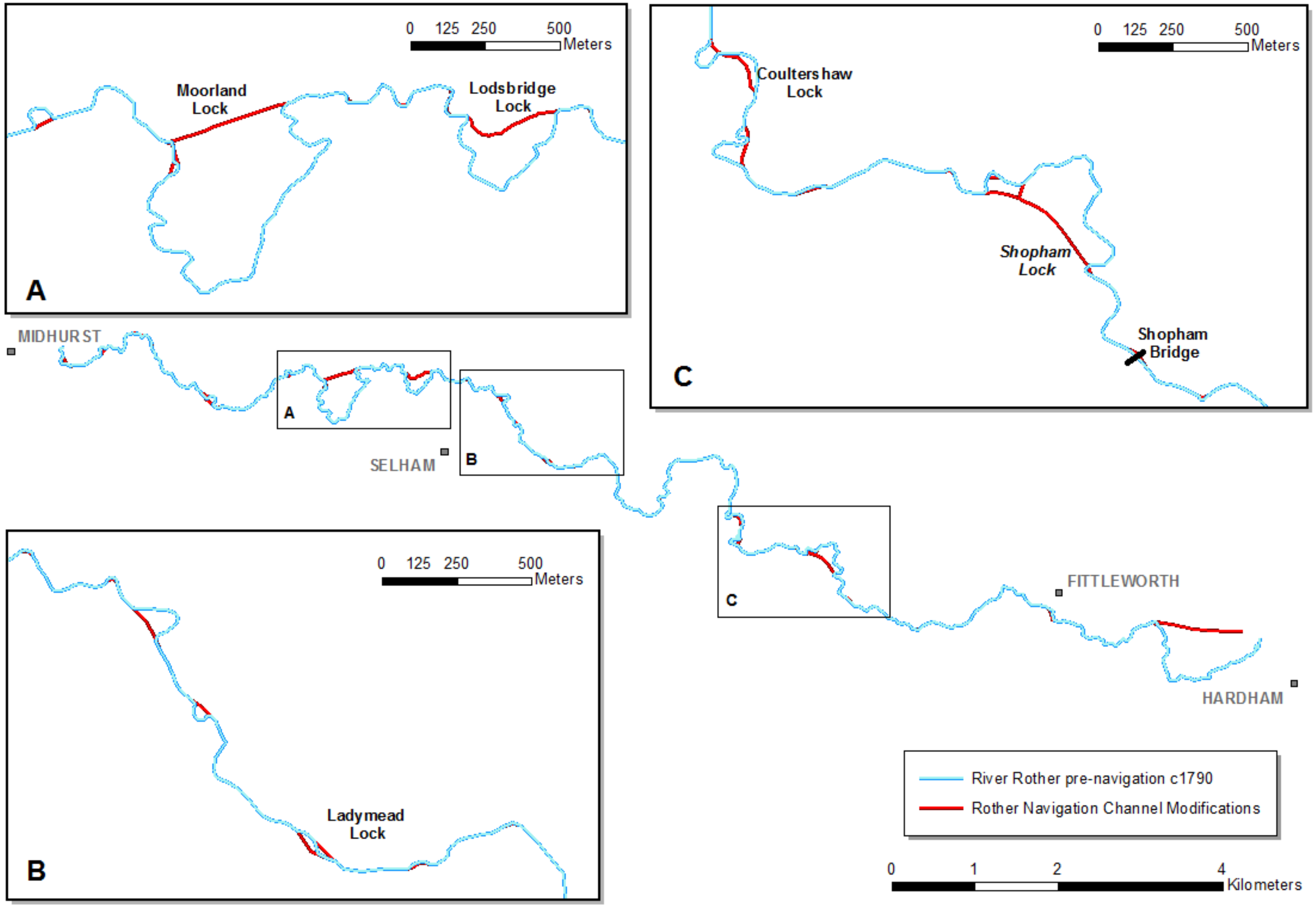


Figure 3.13 Rother Navigation channel modifications and pre-navigation channel c1790, estimated from navigation plans from Petworth Archives

Figure 3.14 Fine sand floodplain deposits on the right bank approximately 30m from main channel in 2014 (photograph taken looking upstream towards Shopham Bridge).

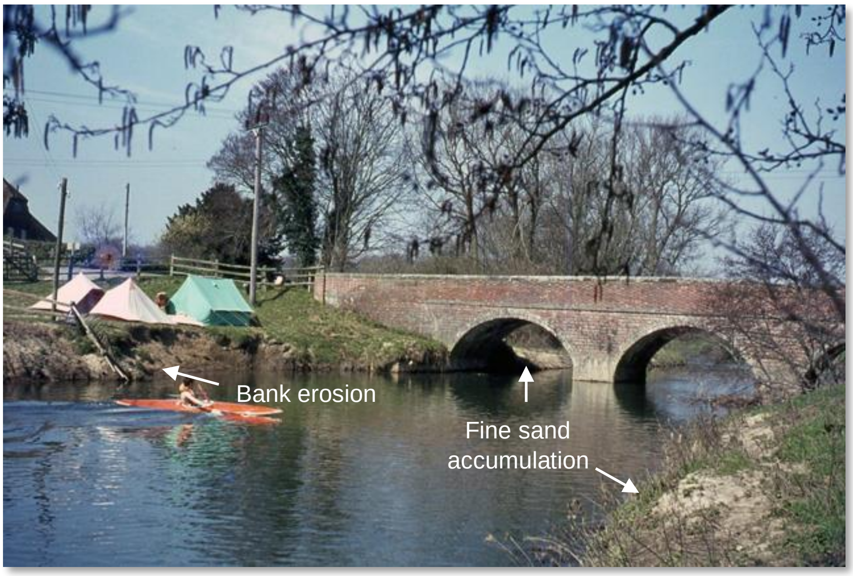


Figure 3.15 'Camping and Canoeing at Shopham Bridge' by Ian Taylor, Source; www.geograph.co.uk. Fine sand deposits in 1969 on left bank downstream and right bank upstream of Shopham Bridge respectively and bank erosion on right bank. Photograph taken looking upstream looking towards Shopham Bridge.

Substantial floodplain deposits were observed following both the 1968 and 2013/14 flood events (Figs 3.14 and 3.15). The former further highlighting that fine sediment was a feature prior to the main shift in agricultural land use from pasture to arable farming in the 1970s as interpreted by Sear (1996). These observed deposits on the floodplain demonstrate that it may still be inundated during very high flow events.

In summary, fine sand accumulation in the River Rother is not a new observation. Several possible factors are identified as contributing to the sandy substrate that is dominant in the lower catchment, namely; a tapering slope, barriers to sediment transport, land-use changes and channel modification. The sheer number of barriers along the channel indicates that if gravel could be adequately sourced from the catchment, the transport of this material downstream to the lower reaches would be highly unlikely. Consequently, in the short-term, the only feasible method to introduce gravel substrate suitable for spawning brown trout into the lower system would be through regular augmentation. However, an overwide channel geometry and lack of lateral connectivity with the floodplain in the lower catchment is likely to promote in-channel fine sediment deposition.

3.2.5 Ecological Status

The River Rother plays a significant role in the maintenance of valuable ecosystems found along the River Arun. However, the waterbody was classified as having an overall ‘poor’ ecological status in 2009 according to EU WFD standards and more recently in 2015, the ecological status was classified as ‘moderate’ (EA, 2017). The assessment of the River Rother includes 11 separate water bodies, of which 2 were classified as poor ecological status and 9 were classified as moderate ecological status in 2015 (EA, 2017). The assessment of the River Arun includes 40 separate water bodies, of which 3 were classified as bad, 13 as poor, 22 as moderate and 2 as good (EA, 2017). Therefore, the ecological status of this river is performing moderately well with respect to the wider catchment.

The chemical status of the River Rother has been classified as good (EA, 2017) which suggests physical habitat modification is a primary factor in the decline of the ecological community. Physical habitat degradation is thought to have resulted from both the smothering of spawning gravel habitat by fine sediment from agricultural run-off and a lack of habitat heterogeneity due to channel modifications (EA, 2002; 2013, Boardman et al., 2009). However, the River Rother may have been a poor fishery for much longer than previously perceived by stakeholders, as archival evidence indicates poor fisheries at least since the construction of the Rother Navigation (Cox and Soar, 2017).

Electrofishing surveys conducted in 2013 and 2014 by the Environment Agency documented eleven major species to be present along the River Rother (Fig. 3.16). The total density of fish decreased with distance downstream, with Terwick Mill the highest quality site, both in terms of density and abundance. In both the upper catchment sampling locations at Standbridge, brown trout and grayling were abundant. Neither species were documented in downstream sampling locations despite stocking of brown trout in downstream reaches by local land managers (M. Williams, Personal Communication, August 18, 2015; A. Thomson, August 12, 2015). It is possible that other limiting factors such as a lack of gravel substrate, shade and refuge habitat in the downstream reaches may have influenced the distribution of fish species.

Other species of ecological importance have been observed within the catchment, including the native White-Clawed crayfish (*Austropotamobius pallipes).* A population recorded upstream of Durford Mill, Petersfield was thought to be the last population in West Sussex separated from Signal crayfish (*Pacifastacus leniusculus)* by in-channel barriers. Signal crayfish have been present in the lower reaches since at least the 1990's (Peay, 2001). As an invasive species, this habitat overlap could be detrimental to the local white-clawed crayfish population, as Signal crayfish can spread crayfish plague (Alderman et al., 1990). However, Signal crayfish have since been discovered upstream of Durford Mill (EA, 2014), which may indicate the foreseeable loss of the White-Clawed crayfish population in the catchment without further intervention (Peay, 2001). In addition, physical characteristics of the catchment may also be altered by Signal crayfish. In other catchments, they have been linked to bank erosion, increased fine suspended sediment concentrations through burrowing activities (Harvey et al., 2014, Rice et al., 2016) and increased sediment transport rates particularly during low summer flows (Johnson et al., 2014).

The Eurasian otter (*Lutra lutra*) is also a valuable indicator of ecosystem health as they are a top predator in the food chain (Peterson and Schulte, 2016). The Eurasian otter had a limited presence along the River Rother in 2011 (King, 2011) indicating limited ecological functioning. The siltation of holt (habitat), lack of riparian vegetation and poor fisheries are all potential limiting factors that could be hindering the recovery of the otter population (Mason and McDonald, 2004). However, low otter numbers may also be influenced by the presence of the American Mink (*Neovision vison)* in the catchment, an invasive species linked to the decline of native species, through both competition and predation (Bonesi and Palazon, 2007).

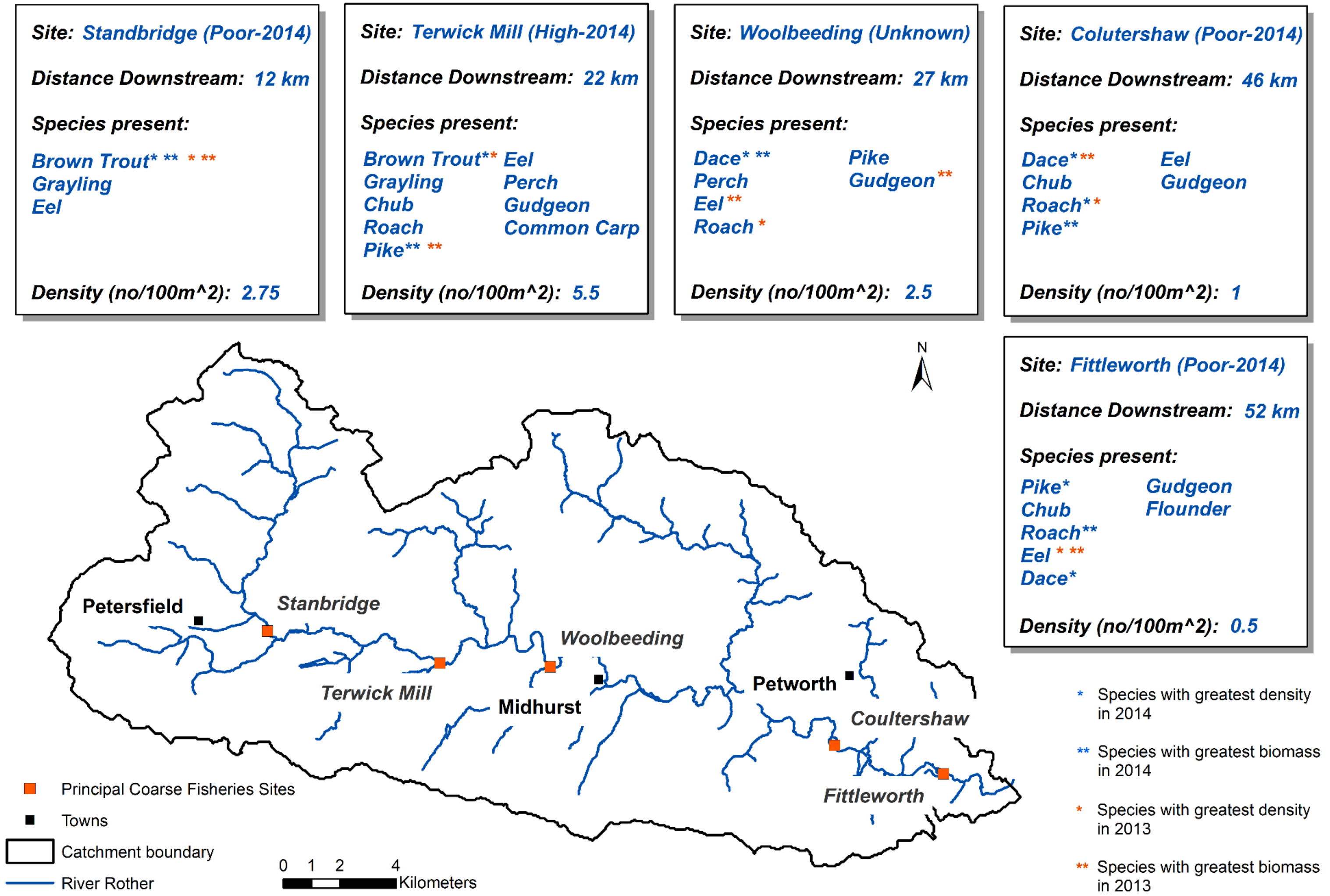


Figure 3.16 Coarse fisheries monitoring along the River Rother (a summary of 2013 and 2014 reports by the Environment Agency for the River Rother catchment).

In summary, the poor 'ecological' status of the River Rother has been a direct consequence of anthropogenic influences within the catchment, not only over the last few decades but centuries (Fig. 3.17). Of significance are channel modifications and shifting land use patterns which have affected channel sediment composition and transport processes over a millennium. Fine sediment substrate in the River Rother is to be expected given the underlying geology of the catchment. However, increased soil erosion resulting from changing agricultural practices in the twentieth century may have been a compounding issue and further degraded the riverine ecological community. The changes in floodplain land use, invasive species, a heavily modified channel and changing hydrology may have all contributed to the decline in the ecological status of the River Rother.

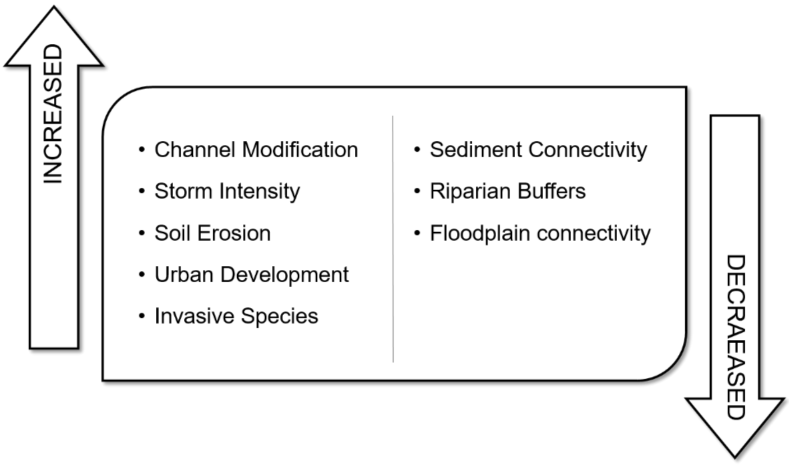


Figure 3.17 Factors influencing ecological quality on the River Rother

3.3 Catchment Initiatives

The impetus for river restoration to improve the ecological and geomorphological functioning of the River Rother is clear from Section 3.2. There is also an economic justification for restoration, as fine sediment is hindering water abstraction by Southern Water at Hardham, near the confluence with the River Arun (Cox and Soar, 2017). The EU WFD has been a primary driver of river restoration in the River Rother catchment as it has been designated as having a 'poor' ecological status. Practices harmful to the river ecology cannot be stopped immediately as land use within the catchment supports a wide range of economically important activities. Nonetheless, in recent years, many local stakeholders (Fig. 3.18) have sought to manage these activities in an environmentally sympathetic way.

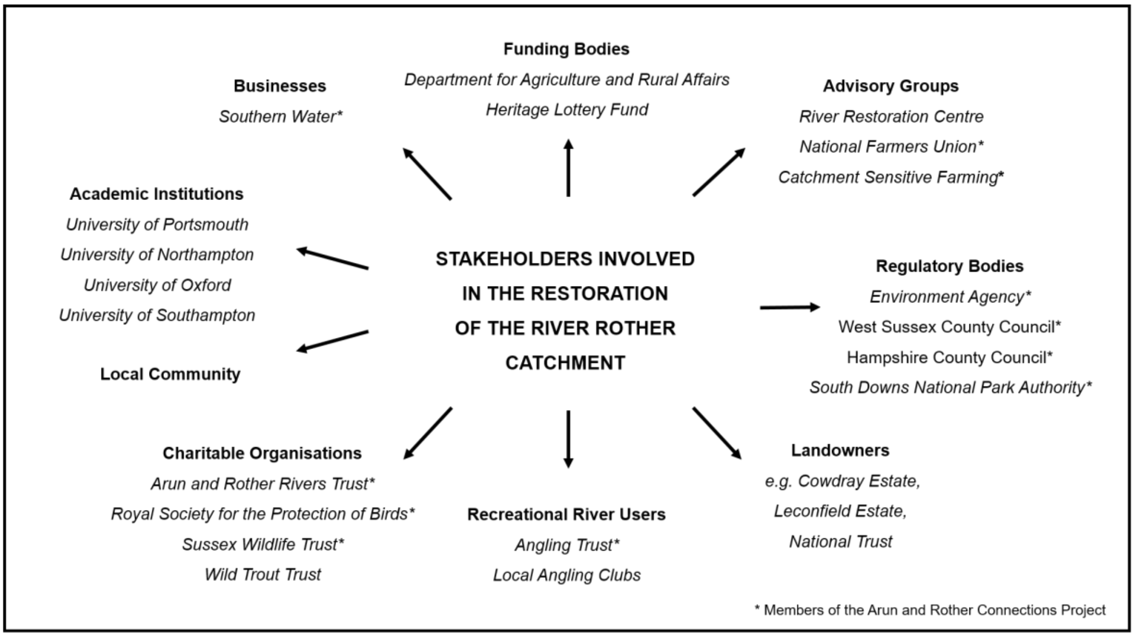


Figure 3.18 Primary stakeholders in the restoration of the River Rother Catchment

River restoration within the catchment has been undertaken over the last 20 years (see Table 3.3). Both reach and catchment scale restoration projects have been enabled over this period, and engagement activities have become a larger feature of restoration projects at both scales. This has increased the number and range of stakeholders involved in restoration initiatives. Academic research projects have also noticeably increased since 2013 (see Table 3.3). Consequently, the future restoration of the River Rother should be informed by an improved scientific understanding of catchment processes and the risk of failure in future restoration projects should be reduced.

There has been a mixture of restoration activities implemented within the catchment, both in-channel (e.g. Shopham Loop and Tilmore Brook restoration projects) and out-of-channel (e.g. in-field sediment traps). More recently, there has been a focus on soil erosion and out-of-channel sediment management measures which aim to treat fine sediment issues at the source. Sediment traps have been trialled within the catchment to reduce the sediment inputs from agricultural run-off, mixed results have been reported but such techniques may be efficient to trap coarser sediments (Wright and Foster, 2014). However, section 3.2 highlights that the restoration of in-channel processes may be just as important for restoring the ecological and geomorphological functioning of the River Rother.

Table 3.3 Examples of river restoration initiatives undertaken in the River Rother catchment.

Initiative	Date	Main Stakeholders	Brief Description
Reach Scale			
Tilmore Brook restoration	2003	RRC, Gifford	Range of in-channel restoration techniques including realignment, backfilling and installation of step pools to mitigate erosion in an incising channel.
Shopham Loop restoration	2004	EA, RRC, UoS	Range of in-channel and floodplain restoration techniques to establish reconnection of a remnant meander (blocked by siltation as a result of the Rother Navigation).
River Rother habitat enhancement scheme	2013-2014	ARRT	Range of in-channel techniques in the Shopham area to improve habitat for brown trout (see section 3.4).
Engaging with river restoration project	2013 - 2015	UoP, SDNPA	Monitoring program (forming the demonstration site of this thesis) and public engagement with wider community to raise awareness of restoration in the National Park.
Fittleworth Mill fish easement	Proposed	ARRT	Modification of barrier at Fittleworth Mill to ease migration of fish.
Catchment Scale			
Rother Valley landcare project	1999 - 2005	Defra	Public engagement with farmers to assess risk and develop management plans to reduce soil erosion.
Arun and Rother connections Project	2013 - 2016	See Fig. 3.1	Catchment scale project (River Arun) aimed at raising awareness and engaging community with catchment restoration, as well as delivering targeted restoration projects.
SMART Project	2013 - 2016	UoN, UoO, SDNPA	Catchment scale investigation using sediment fingerprinting techniques to identify potential transport routes of agricultural run-off within the catchment.
A-STAR Project	2015-2017	UoP, SDNPA, EA, SW	Catchment scale investigation into in-channel sediment issues using broadscale modelling.
Forgotten Fields	2016-2018	UoP SDNPA, National Trust	Catchment scale investigation into land use change over the last 150 years using 19th Century Tithe Maps to inform catchment scale restoration.
Payment for ecosystem services project	Proposed	ARRT, SW, UoC, SDNPA, EA	Catchment scale project using outputs of the SMART and A-STAR projects to restore ecosystem services by local landowners funded by beneficiaries (e.g. Southern Water).

3.4 River Rother Habitat Enhancement Scheme

In 2013, the Arun and Rother Rivers Trust (ARRT) secured approximately £110 000 from the Catchment Restoration Fund (CRF; Defra, 2014) to focus on fisheries habitat enhancement along the River Rother near Shopham Bridge, Petworth (Fig. 3.1). The River Rother Habitat Enhancement Scheme (RHES) focused on a section of the River Rother stretching 5.6km in the lower catchment between Coutlershaw Bridge and Fittleworth Mill (Figure 3.14). Fish and molluscan surveys undertaken along this stretch prior to restoration reiterated the poor ecological status seen elsewhere in the catchment (Willing, 2014). Both Coultershaw Bridge and Fittleworth Mill have historically imposed significant barriers to sediment transport and migratory species, and thus, a threat to habitat quality as they fragment habitat beyond the reach scale. Fish easement programmes to ease migration have been undertaken at both structures, the programme at Fittleworth finished in 2016 (ARRT, 2016). Both structures remain significant barriers to sediment transport (Cox and Soar, 2017).

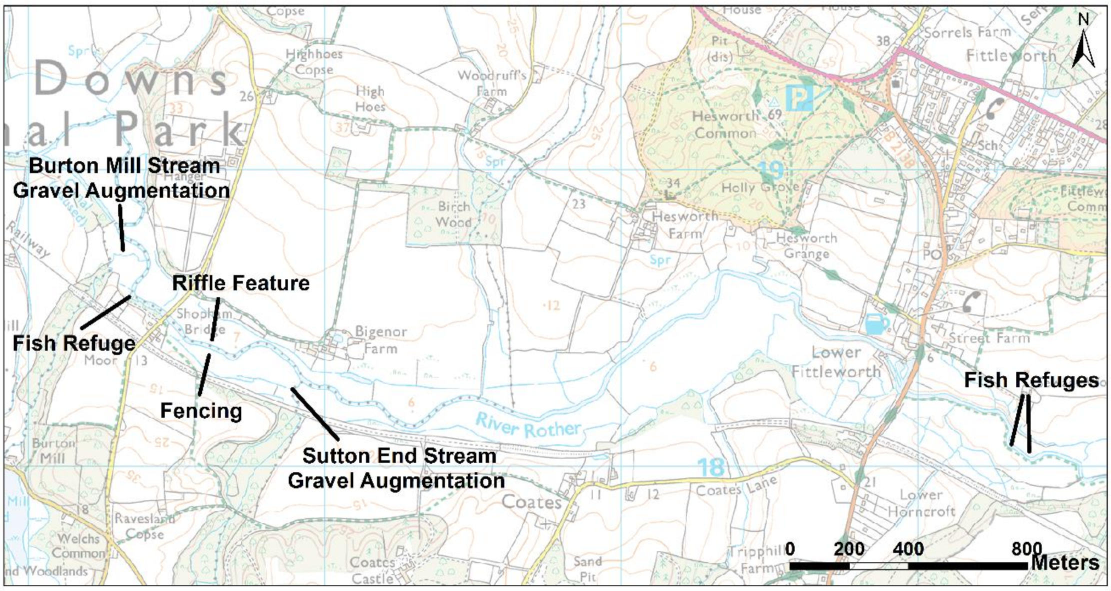


Figure 3.19 Features of the RHES funded by the CRF, and additional features implemented following the completion of the main body of works proposed in the CRF application.

The RHES incorporated four elements of habitat enhancement: gravel placements on two tributaries, namely Sutton End Stream and Burton Mill Stream; a fish refuge 150m upstream of Shopham Bridge; and a 60m artificial riffle feature in the main channel downstream of Shopham Bridge (Fig. 3.14). The latter was intended to provide spawning habitat for coarse fish, particularly for brown trout, which were found in low abundance in the lower catchment (Section 3.2.5.). This riffle feature is the focus of the exploratory monitoring programme undertaken as part of this research study.

In addition to a lack of spawning habitat, poor habitat quality in the reach was also compounded by a lack of riparian vegetation and thus, shade. The only two Alder trees observed within the reach prior to restoration, and these were at a high risk from the Phytophthora disease. In addition, the riparian community had been directly impacted by farming on the adjacent fields to the channel. Both historical and recent poaching of the banks by cattle was evident in fields used for pasture (Fig. 3.20). Therefore, restoration work beyond that of the riffle feature was required to improve habitat quality in the reach. Initial action had been taken by the landowner to fence the left bank prior to restoration. The CRF restoration work described above were all implemented in 2013. In 2014, additional riparian restoration was undertaken, specifically fencing and tree planting programs intended to re-establish a riparian community along the riffle feature (Fig. 3.21).

Figure 3.20 'Time for a paddle in the River Rother' (2009) taken looking downstream near Shopham Bridge. Source Dave Spicer, www. geograph.org.uk

Figure 3.21 Fencing erected and tree planting in 2014 preventing cattle from accessing the riffle feature. The fencing incorporated stiles to retain access for local anglers.

Figure 3.22 Images of the RHES through the phases of construction. Top left – Before restoration in July 2013. Top right – During construction of the riffle feature. Middle left – immediately following construction of the riffle feature. Middle right – immediately following construction of the riffle feature (riffle tail). Bottom right – 1 year post-construction of the riffle feature.

Riffles are in-channel topographical highs commonly observed in gravel-bed rivers as part of a pool (topographical lows) riffle sequence (Wohl, 2014). Many theories have been developed on the maintenance of pool riffle sequences, including the velocity reversal hypothesis (Keller, 1972) and the flow convergence routing hypothesis (MacWilliams, 2006). However, these theories are developed around sediment transport and assume a constant supply of coarse sediment. Gravel is observed in some areas upstream of Midhurst but the transport of this material downstream to the RHES site is impeded by several large instream barriers. Based on the evidence reported in section 3.2, a gravel augmentation scheme such as the RHES was the only likely way of achieving gravel spawning areas for brown trout in the short term.

Spawning habitat enhancement, such as this riffle feature, can provide immediately usable habitat but must be considered in the appropriate context of geomorphological processes to ensure longevity (Wheaton et al., 2004a; 2004b). The River Rother has been perceived as being a sandy-gravel river historically (Sear, 1996; Boardman et al., 2009) but now, through the modification of catchment processes, it exhibits geomorphological forms more closely associated with a sand bed river. The restoration of pool riffle sequences in this area of the catchment has been undertaken as an ‘enhancement’ as suggested in the project title.

Enhancement has been defined as either “any improvement in environmental quality” (Brookes and Shields, 1996), or “any improvement to a structural or functional attribute” (National Research Council, 1992). The latter definition would assume riffles were a feature once present within the reach. However, the new evidence reported here and in Cox and Soar (2017) casts doubt on the historical extent of pool-riffle forms in the lower catchment. This was not available at the project outset but could provide supporting evidence for future restoration activities. Therefore, creation, ‘bringing into being a new ecosystem that did not previously exist at the site’ (National Research Council, 1992), may be a more appropriate term for this form of restoration.

The construction of the riffle feature began on the 1st August 2013 and was completed within a week (Fig. 3.22). The feature was designed (in collaboration with the Wild Trout Trust) to be stable at bankfull flows. It comprised a mudstone base (for stability) which was overlain with a gravel substrate. The substrate added to the channel during construction consisted of mainly very coarse to coarse pebble gravel, and was significantly different from the pre-restoration coarse sand substrate (Fig. 3.23). The gravel substrate was placed in the channel to increase the height of the feature (reduce water depth) from the head of the feature in the downstream direction. The feature was designed to have a maximum elevation (shallowest depth) at the tail, which would break the water surface (Fig. 3.20).

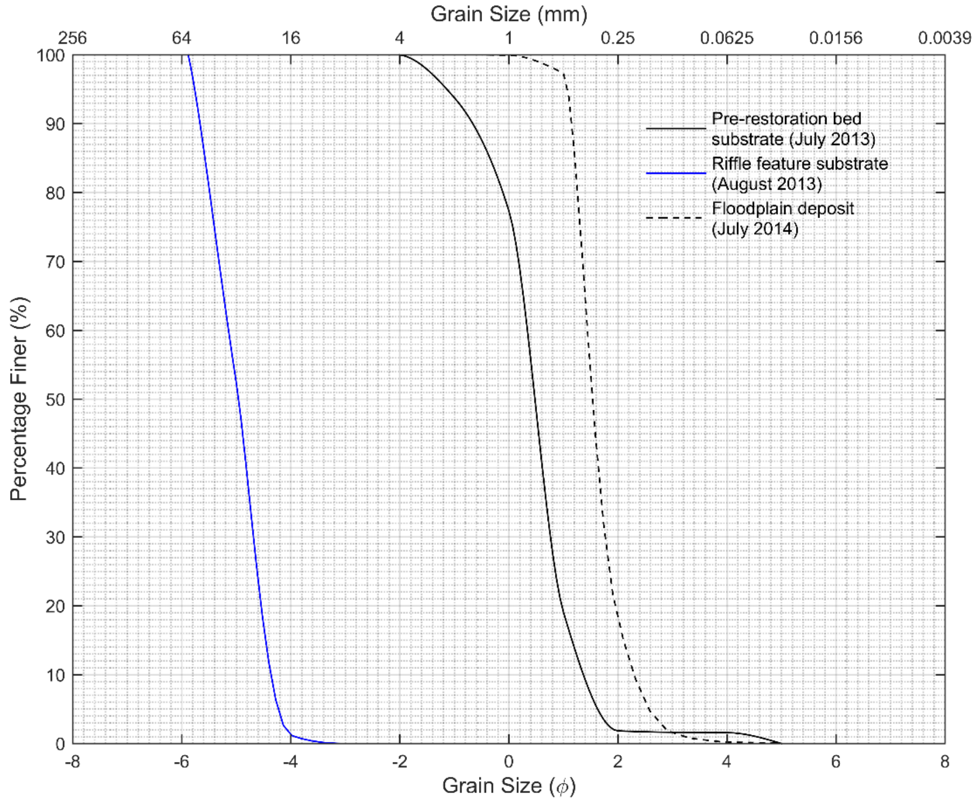


Figure 3.23 Particle size distribution of sediments sampled from the bed prior to restoration, the artificial material added to the river to form the riffle and deposits on the floodplain from the 2013/14 winter floods

Fine sediment storage in the channel is highly likely as discussed previously in this chapter, therefore, the risk for substantial sedimentation of augmented gravel in this environment was high. Gravel augmentation to form of riffles in the area has shown mixed results. Bed check riffles constructed in the Shopham Loop restoration scheme do appear to remain functional on visual inspection, whilst other riffles have required replenishment since initial restoration activities were undertaken. Not all of these features were monitored and the data from those that were monitored were insufficient to understand the processes maintaining these features. Limited opportunities had been available to inform the design of the River Rother Habitat Enhancement Scheme. It was uncertain if there would have been sufficient material over the long-term to maintain the coarse fraction of the riffle feature. Therefore, the uncertainty surrounding the longevity of the scheme was an impetus for a data-driven performance assessment.

In natural riffles, the riffle head acts as a foci for sediment deposition as flow diverges upon exit of an upstream pool and water depths decrease. Accordingly, sediment tends to fine towards the tail of the riffle due to selective entrainment. The riffle feature discussed herein was designed and constructed inversely to naturally occurring riffles observed in gravel bed rivers. Although the ARRT monitoring programme of the RHES included invertebrate and fish surveys as well as repeat fixed point photography, no direct geomorphological monitoring was planned. This was, however, seen as a unique learning opportunity by the University of Portsmouth and South Downs National Park Authority. The impetus to monitor this feature was therefore threefold:

i) To assess if the riffle feature retains structural integrity as designed, given the atypical and unique structural design of the feature;
ii) To assess if the feature has been successful in terms of promoting sediment transport and resisting extensive aggradation of fine sediment; and,
iii) To inform interested parties of the performance of the feature (both positive and negative) thus ensuring the necessary data and knowledge is available to inform future restoration work.

3.5 Summary

Undertaking a comprehensive review of available catchment datasets is critical in designing and evaluating the performance of river restoration schemes (Downs and Kondolf, 2002; Downs et al., 2011; Friberg et al., 2016). A review of the River Rother catchment was not undertaken prior to the design and implementation of the RHES. It is conceivable that had this type of review been undertaken at the outset of the RHES a different design approach may have been adopted. However, the project was financially limited and gravel augmentation in the lower reaches does appear to be the only feasible method for creating spawning habitat within the lower catchment over the short term.

This chapter has challenged some pre-conceived perceptions of the drivers of physical habitat degradation within the River Rother catchment. The information presented within this chapter, along with other studies, may be used by stakeholders to guide future river restoration aims and SMART objectives. This review has been achieved mostly through using open data sources which are increasingly becoming available. Some material used within this review were highly site specific, such as the information gleaned from a particularly rich local archive. However, in general this type of study could be easily replicated at a low cost in the UK, and beyond if the datasets are widely available. This chapter also serves to highlight the importance for continually reviewing and considering new datasets that become available. Although a comprehensive review of the catchment was undertaken previously in 1996, 20 years later new datasets may lead to different conclusions on catchment scale perceptions. Some new sources of information which may be particularly beneficial for other catchment baseline studies include;

- Local and national archives;
- Cross-sections from existing flood-risk models;
- Open LiDAR and oblique aerial imagery (emergency response);
- Google Earth™;
- A vision of Britain through time website (http://www.visionofbritain.org.uk);
- Britain from Above website (https://britainfromabove.org.uk);
- Art UK website (https://artuk.org);
- The Mills Archive website (https://millsarchive.org) and
- Geograph website (http://www.geograph.org.uk/).

The key findings from reviewing the datasets reported in this chapter will be used to contextually evaluate the geomorphological and physical habitat performance of the RHES in Chapters 5 and 6, respectively. This chapter has outlined the main objective of the RHES. The monitoring programme of the RHES explores existing methods and novel approaches for evaluating geomorphological and physical habitat performance of river restoration. Chapter 4 will outline these approaches based on the review of river restoration monitoring practices discussed Chapter 2.

4 River Rother Habitat Enhancement Scheme: Monitoring methods

4.1 Data Collection

This chapter outlines the exploratory monitoring programme of the River Rother Habitat Enhancement Scheme (RHES) detailed in Chapter 3.4. The review of the emerging technologies in fluvial geomorphology in Chapter 2, identified the ADCP as a potential technology to enhance the learning potential for river restoration monitoring. Therefore, this monitoring protocol was designed to assess the feasibility of using this technology for that purpose. The chapter outlines the data collection, processing and analysis methods used within this research study. The 60m riffle-glide feature of the River Rother Enhancement scheme was monitored intensively within a 180 m reach (Fig. 4.1), over an 18 month period to capture the short-term geomorphological response of the restoration scheme. The spatial extent of the baseline survey, and therefore subsequent surveys, were designed to capture 60m both upstream and downstream of the feature, as the geomorphic adjustment may propagate both upstream and downstream. This length of channel was deemed feasible for a day of surveying during a pilot study undertaken in July 2013 using an ADCP, to minimise the impact of a fluctuating discharge on velocity measurements. The feature was installed 30m upstream from where it was originally planned, therefore, the extent of the monitoring for subsequent surveys extends 30m upstream and 90m downstream.

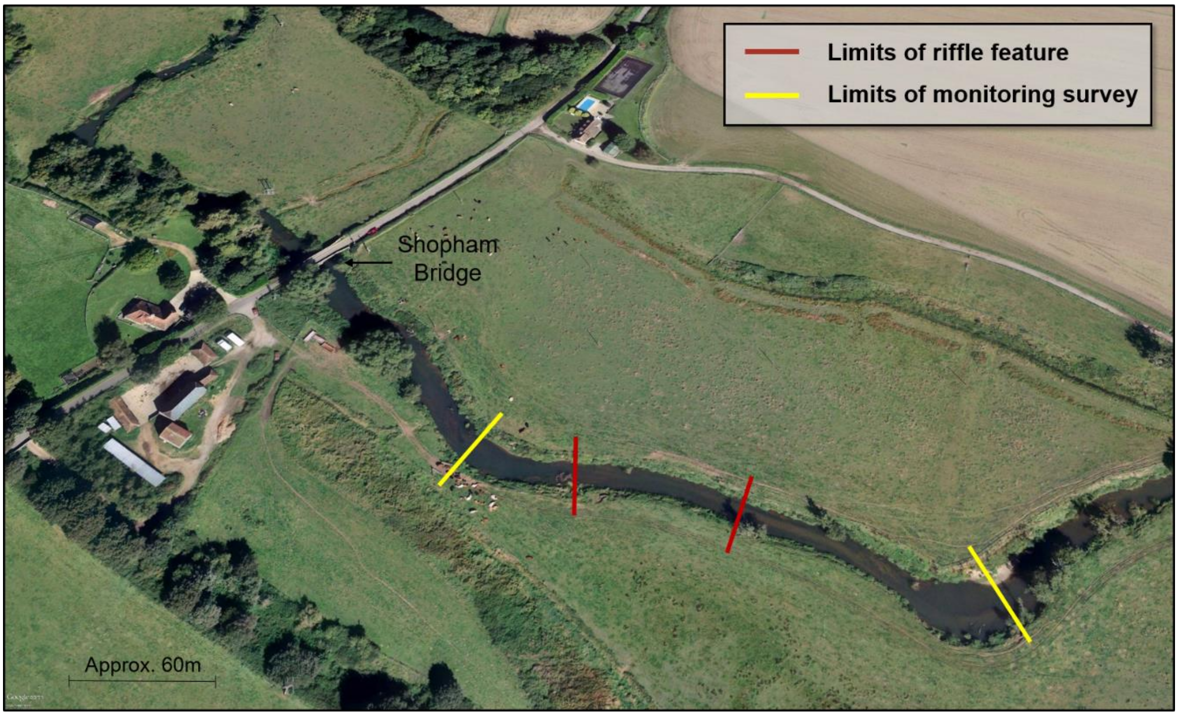


Figure 4.1 Aerial Imagery (Google Earth) identifying limits of the monitoring survey

The scheme was monitored during low flow conditions (1.8 $m^3 s^{-1}$) in July 2013 prior to construction to form the baseline survey. Ideally, other baseline surveys at a range of flows (moderate and high) would have been captured to complement this low flow dataset. However, a short lead in time reduced the pre-project monitoring period available and no suitable moderate or high flow conditions occurred during this period. This short lead in time, and lack of suitable reference sites, also led to the development of a less complex 'Before and After' (BA) monitoring design (Chapter 2). The scheme was surveyed immediately following construction in August 2013 to form the as-built survey and subsequently after 1, 3, 7, 12 and 18-months post-construction to capture any rapid change following construction (Table 4.1). Further monitoring to capture the longer-term project implications (>10 years) is recommended but this is beyond the scope of this project. The 7-month survey was initially planned to be undertaken after 6 months but the site was inaccessible due to flooding (Figure 4.2 and 4.3).

Figure 4.3 The reach (looking downstream) during an out of bank flow of the 2013/14 food events.

Figure 4.2 Image taken from the road on the left side of the channel during flood events highlighting the extent of the flooding in February 2014.

Table 4.1 Details of surveys undertaken using the ADCP as part of the monitoring campaign of the RHES

Survey	**Date**	**Discharge ($m^3 s^{-1}$)**	**No of cross-sections**
Baseline	26 / 07 / 2013	1.8	71
As-built	08 / 08 / 2013	1.9	91
1 Month	04 / 09 / 2013	1.7	95
3 Months	09 / 11 / 2013	8.5	76
7 Months	02 / 03/ 2014	13.1	69
12 Months	05 / 08 / 2014	2.1	87
18 Months	24 / 01 / 2015	8.1	92

The Sontek RiverSurveyor S5, Sontek RiverSurveyor M9, Teleydyne StreamPro and Teleydyne RiverRay ADCPs were considered for this project, before proceeding with the Sontek RiverSurveyor S5 ADCP (Fig. 4.4). This ADCP was used to estimate discharge and capture cross-sectional velocity (+/- 0.02 m s^{-1} accuracy) and depth data (99% accurate) (Sontek, 2010). The model was chosen based on profiling range, the functionality of the design and affordability. The S5 has a profiling range of up to 5m for velocity measurements and 20m for depth measurements. This was seen as appropriate given the channel dimensions of the study site; the maximum depth measured during the monitoring period in near bankfull flow conditions was 4.15m. Surveying during out of bank conditions presented significant health and safety risks and as such, surveys were not undertaken in these conditions. Alternative remotely controlled ADCPs such as the ARC-Boat (HR Wallingford, 2016) could have been used during such conditions, but such platforms were prohibitively expensive and not a suitable option for this research study.

As with other non-remote controlled ADCPs with extended profiling capabilities (e.g. M9 and RiverRay), the platform dimensions of remote controlled ADCPs were typically larger. Given the transducer in most ADCP models is mounted in the centre of the equipment, the potential blanking distance (unmeasurable area of the channel) increases at the channel margins with larger models (Fig. 4.4). The minimum blanking distance from the bank in the absence of obstructions using the S5 RiverSurveyor was 23cm. The distance between the centre of the transducer and the bank (including obstructions), is henceforth referred to as the edge distance. Of the smaller ADCP models, the design of the S5 RiverSurveyor appeared the most suitable for data collection during all survey conditions. The transducer was mounted through the centre of the float, whereas the transducer was mounted over the front in the Teledyne StreamPro (Teledyne, 2016). Consequently, there was a risk that the StreamPro could have lost stability during turbulent conditions. The RiverSurveyor S5 presented the most suitable ADCP model for maximising the horizontal extent of the in-channel survey for this monitoring programme.

The RiverSurveyor S5 transducer had a four 3.0 MHz beams in a Janus configuration which was used to collect 3D velocity data. Three components of velocity were measured; easting (V_e), northing (V_n) and vertical (V_z) velocity and were later resolved using Pythagoras theorem. To detect velocity, the RiverSurveyor S5 required water depths of 0.3 m due to the vertical blanking distance. This did limit the measurement of velocity in some marginal areas of the channel, particularly during low flow surveys. Additionally, the transducer had a vertical 1.0 MHz beam which was used to collect depth data. This vertical beam was used to estimate depth in this study over a ‘bottom track’ method which uses an average depth from the four 3.0 MHz beams,

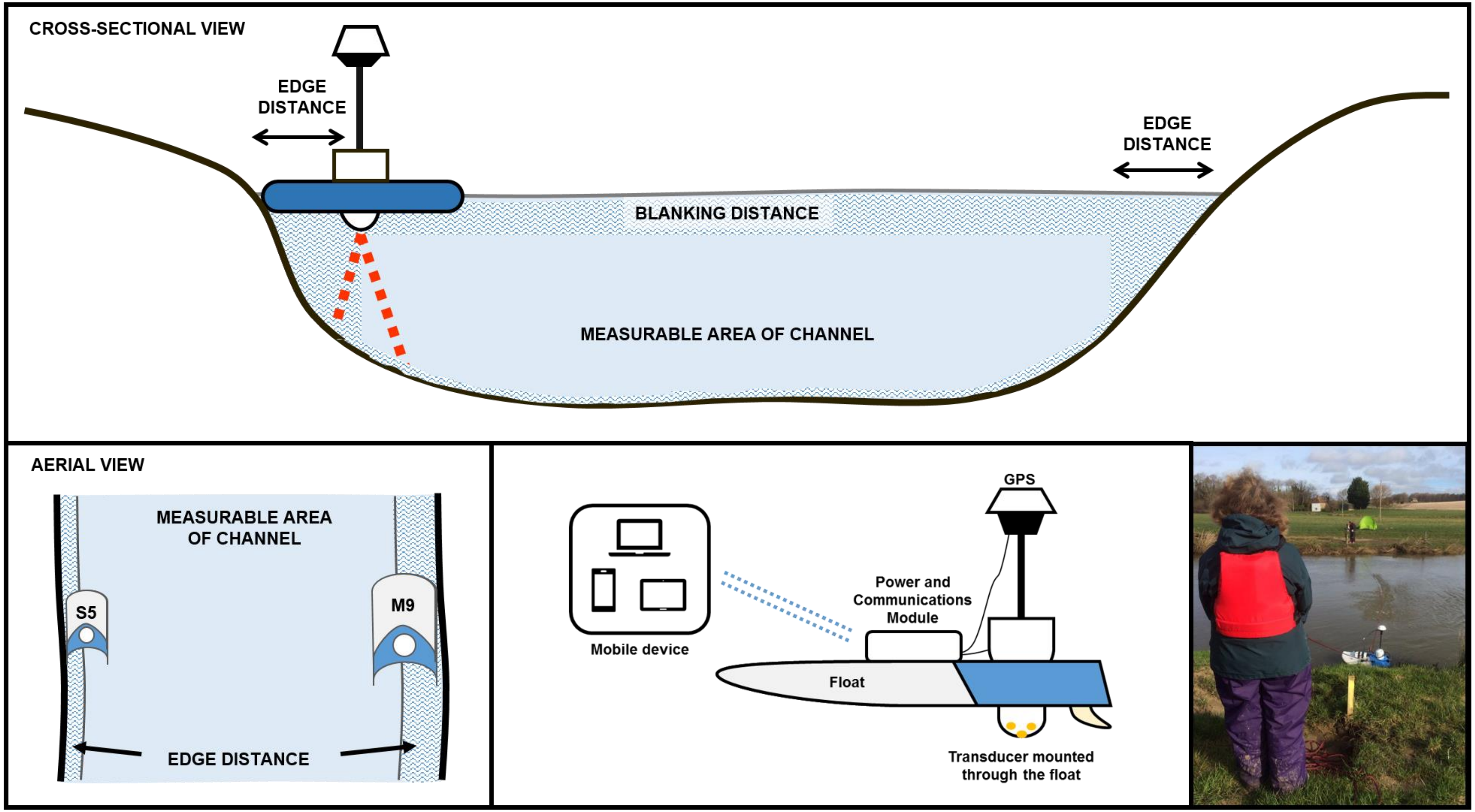


Figure 4.4 Annotated diagrams of the operation of the ADCP specifically the Sontek RiverSurveyor S5 used within this study

as it was viewed as a more robust measurement for identifying geomorphic adjustment.

The transducer was mounted to a floatation device which was pulled from bank to bank along transects at a speed slower than water speed to reduce the risk of error (Fig. 4.4). The transect approach was selected over a random path or zig-zag approach (Williams et al., 2015) so that the edge estimates could be used to identify the extent of the wetted channel area. With the emergence of new technology, the wetted channel area could potentially be delineated using aerial imagery captured using a UAV. This may allow the ADCP to take a random path across the channel, increase the resolution of bank data (using SFM) and reduce ADCP surveying time. Seasonal vegetation could potentially obstruct this method and UAVs were not considered at the time of the baseline survey. Cross-sections were measured on average every 2m throughout the reach, the density of the cross-sections increased over the riffle feature to 1m intervals to ensure any change to the feature over the monitoring period was captured in sufficient detail.

The measured values collected by the ADCP were referenced using SBAS DGPS, with a GGA-GPS string to give sub-metre horizontal accuracy. The GGA-GPS calculates the velocity of the ADCP using two successive measured coordinates, the distance between the points is divided by the time between measurements (Wagner and Mueller, 2011). This reach presented a minimal risk of multipath error to GPS data which has been previously identified as a source of measurement error from structures and vegetation (Rennie and Rainville, 2006). The site was a low risk because the banks were sparsely covered with riparian vegetation, and there are very few local obstacles. Accuracy may be improved with an RTK upgrade to the equipment, but this was not available for this research study due to financial restrictions.

A retrospective assessment of the potential impact of positioning error was undertaken from data collected within this study. In each survey, the first ADCP transect was undertaken from the same location at the top of the reach and an assessment of the variability between points along the surveyed transects was evaluated. Whilst the published GPS accuracy of equipment is suggested as sub-metre, the results of the assessment indicated that the difference in sampling locations ranged from 0.05m to 0.23m. This sampling error is a cautious estimate of the likely error, as these values are also likely to include error resulting from operator positioning on either bank, the tautness of the rope and other environmental conditions such as wind and water velocity. Given the post-processing protocol (Section 4.2) spatially averaged the data points within a 1m radius of a grid node, this error is unlikely to have had a significant impact on the results presented in this study.

The GGA-GPS string was preferred over the bottom-tracking for referencing ADCP measurements as a moving-bed was detected. Studies conducted in moving bed conditions using bottom tracking have consequently noted that the ADCP underestimates velocity measurements (Mueller and Wagner, 2007; Wagner and Mueller, 2011). Methods are available to correct for a moving bed and thus avoiding the risk of multipath error, such as the Loop Correction method (Mueller and Wagner, 2007). However, this method requires the ADCP to profile the targeted cross-section twice in the same transect (i.e. to complete a 'loop'). If this approach was adopted, the surveying time would have doubled and may have negated the expediency of the data collection using an ADCP.

The frequency of cross-sections decreased when surveying during higher flows, because the time taken to survey each cross-section increased due to a larger wetted channel width. However, the data collected during higher flows was still of a high resolution. In contrast, seasonal narrowing of the channel due riparian vegetation reduced the cross-sectional surveying time and hence increased cross-section frequency during summer low flows. An ADCP was also used to estimate discharge, in addition to capturing cross-sectional velocity and depth data throughout the reach. ADCP depth data was later rectified to ordnance datum using water surface elevation data which was surveyed along the reach with a TopCon HiperPro RTK GPS (+/- 0.015m accuracy). The RTK GPS was also used to capture bank elevation data at a 1-0.5m resolution (top and bottom) and floodplain elevation data at a 10m resolution. Each ADCP survey was completed within a day, with an additional day for surveying change along the banklines.

Substrate was identified as an important physical habitat parameter (Chapter 3), and thus attempts to quantify substrate were made during the monitoring programme. However, these attempts were unsuccessful as some areas of the study reach were too deep to enable an adequate spatial coverage even during low flow conditions. In wadeable conditions, the sampling process was excessively time-consuming. This was not considered repeatable for a typical river restoration monitoring project given the available measurement techniques at the time of this monitoring programme. Given that physical habitat studies have been successful using only velocity and depth parameters (Emery et al., 2003, Brown and Pasternack, 2009, Wallis et al., 2012), substrate sampling was limited to spot samples over the riffle feature. Samples were taken during low flow surveys, prior to and immediately following restoration and 12-months post construction of the riffle feature. The particle size distributions of the samples were obtained through dry sieving following standard laboratory procedures using a stack of sieves with apertures between 32 mm-63 μm (British Standards, 1990). The potential for fine sediment storage within the riffle feature was monitored as part of a wider monitoring campaign in a separate research project (see Evans et al., 2017).

Other methods have been used to quantify substrate, such as high-resolution aerial imagery (Bergeron and Carbonneau, 2012), Terrestrial Laser Scanner (TLS, Entwistle and Fuller, 2009) and Structure from Motion (SFM, Woodget and Austrums, 2017). However, these methods were not appropriate due to high turbidity levels at the site (as a result of suspended sediment). The ADCP has also been applied to classify substrate using backscatter intensity values but the study found difficulty in distinguishing between gravel and sand in their study sites (Shields, 2009). Given the nature of the pre-restoration substrate (sand) and the substrate used during restoration (gravel) at the Shopham Bridge reach, this application of the ADCP was not deemed viable in this environment. Further research and development in using the ADCP for this purpose is promising for physical habitat assessments but is beyond the scope of this study.

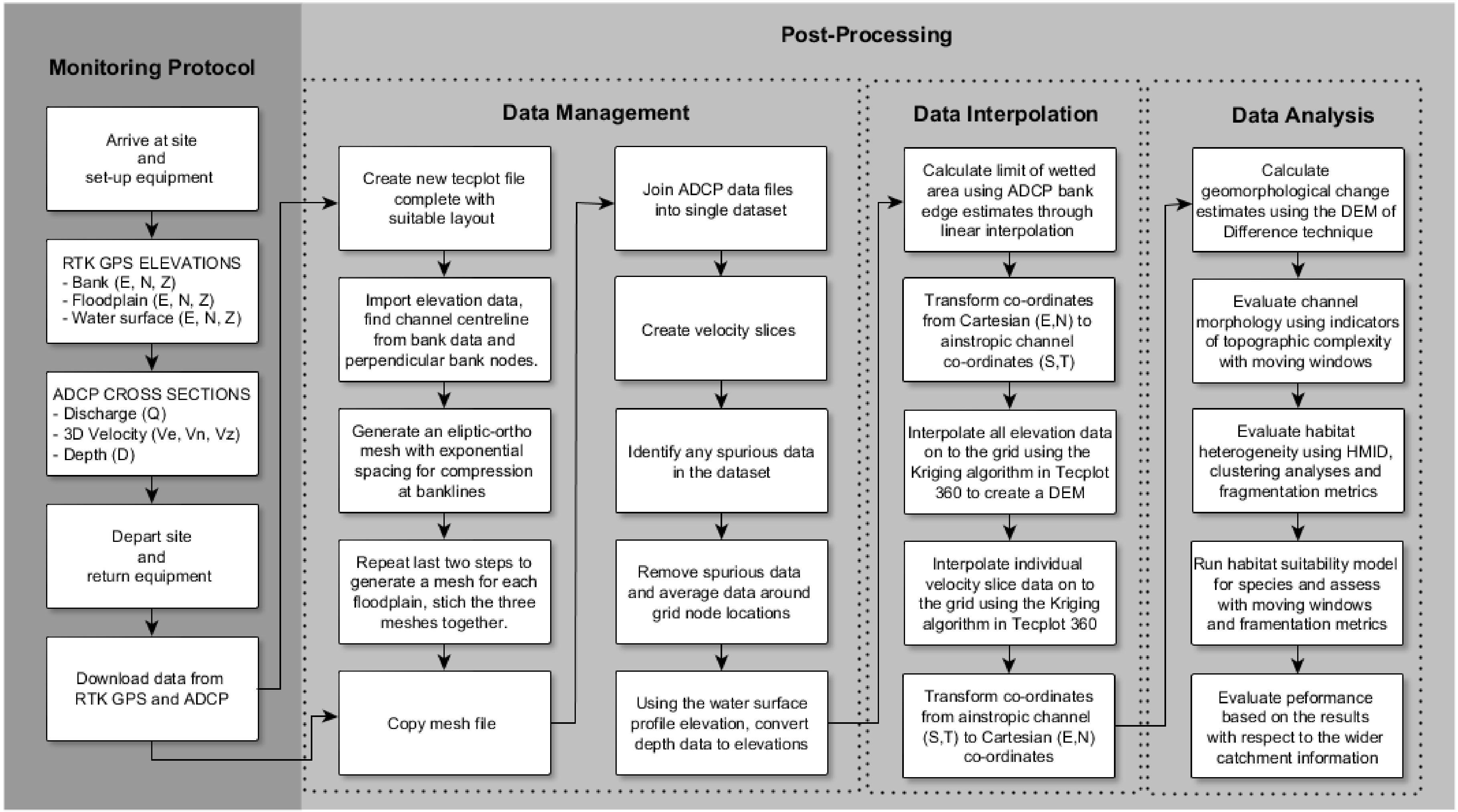


Figure 4.6 Workflow of the monitoring programme of the RHES reported within this chapter. Summary of steps for data collection, data management, data interpolation and data analysis

4.2 Data Processing

Existing software such as the Velocity Mapping Toolbox (VMT) (Parsons et al., 2013) was accessible for the post-processing of ADCP data. However, limited support for Sontek ADCPs was available at the time of this research project but has been recently developed (USGS, 2017). The development of a post-processing protocol (Fig. 4.6), using both Matlab scripts and Microsoft Excel Macros, was required to extract, transform and initially analyse the data. The post-processing protocol is divided into two sections; data processing and data interpolation.

4.2.1 Post-Processing

This section of the post-processing protocol explores the process of transforming the field data into a usable format for subsequent analysis (Fig. 4.7). In the first instance, a structured grid was created in Tecplot 360 (Tecplot 360, Bellevue, 2011), using the bank and floodplain data collected during the baseline survey. The grid was consistent for each survey. A structured grid was adopted over a Triangle Irregular Network (TIN) to provide consistency between data points for subsequent analysis of the variables. The use of a structured grid addressed the imbalance between excess data in the cross-stream and a paucity of data in the downstream. This approach enabled the survey data to be smoothed around the grid node locations and alleviated the inconsistency in sampling locations between surveys. The grid was created with an average cell size of 1m along the channel centreline in accordance with the minimum transect sampling frequency (Fig. 4.8). The use of an elliptic-ortho grid permitted cell sizes to compress near the banklines to account for any rapid change in the margins, whilst preserving orthogonality and maintaining a consistent cell size.

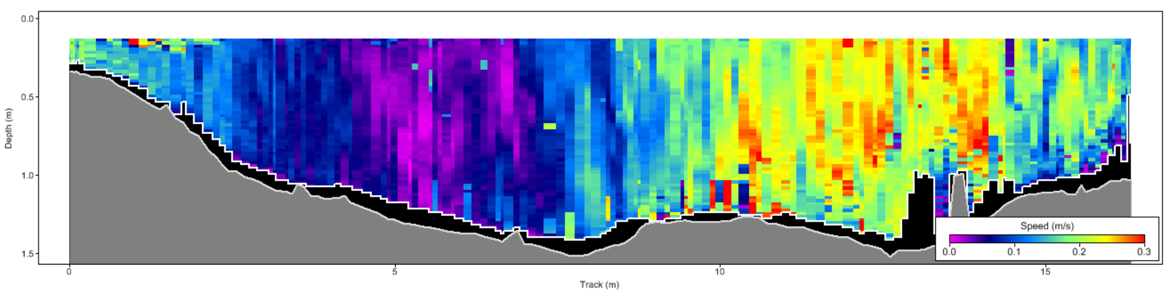


Figure 4.7 An example of a cross-section from RiverSurveyor Live showing the high resolution of the raw data

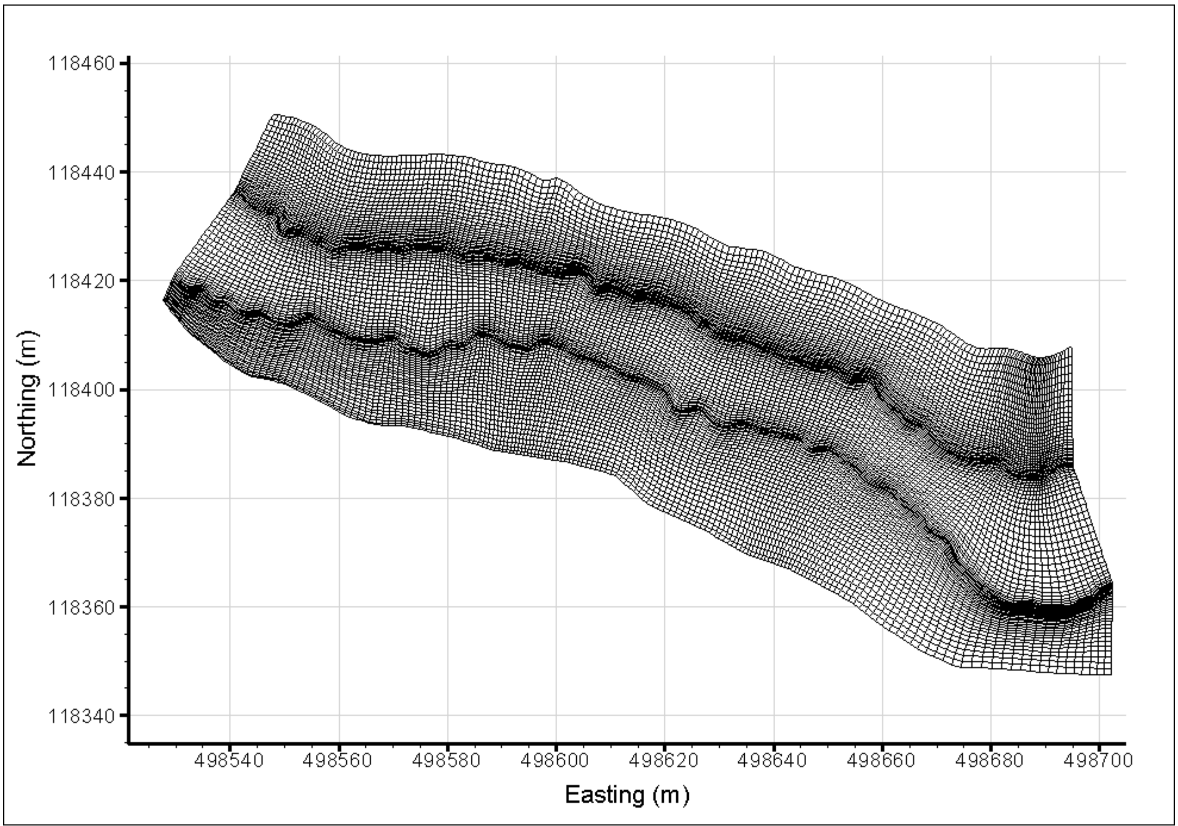


Figure 4.8 Structured grid of reach downstream of Shopham Bridge restored in 2013 under the Catchment Restoration Fund

The ADCP data are available in real-time via radio or Bluetooth but the data are also stored onboard the transducer within the Sontek S5 RiverSurveyor ADCP. This minimised the of data loss through wireless transmission but also required the data to be downloaded from the equipment. Once downloaded, preliminary post-processing of the data was performed in RiverSurveyor Live (RSL). This post-processing included the correction of settings applied in the field and the calculation of discharge estimates. The discharge estimate for each survey was approximated from an average of repeated cross-sectional measurements collected throughout each survey. Further post-processing of velocity and depth measurements was performed outside of RSL, and cross-sectional data was exported from the software as both as ASCII (.vel) and Matlab (.mat) files. The 3D velocity component data was only stored within the .vel file. This was extracted to Matlab to create a master data set of velocity and depth data with their associated coordinates.

The ability of the ADCP to measure data at a high resolution (2cm bins), resulted in the generation of more velocity data than necessary for the application of river restoration monitoring, particularly in the vertical dimension of the cross-section (Fig. 4.9). Emery et al. (2003) used only the depth average velocity measured at 40% of the depth above the bed. This is a standard point measurement procedure to account for the logarithmic velocity profile in a shallow environment. More recently, applications of the ADCP for geospatial representation of velocity have also used the depth average velocity (Rennie and Church, 2010; Jamieson et al., 2013; Williams et al., 2015). However, this study adopted a different approach similar to Tsubaki et al. (2012) to utilise the high resolution data, and simultaneously address the imbalance between the vertical, cross-stream and downstream data resolution. Five velocity slices were adopted at 20%, 40%, 60%, 80% and 95% (near surface) of the depth above the bed, to retain an appropriate resolution in both deep and shallow areas of the surveyed channel. The velocity for each slice is derived from the average of the data 10% above and below the velocity slice height, and 5% above and below for the near surface slice (Fig. 4.9).

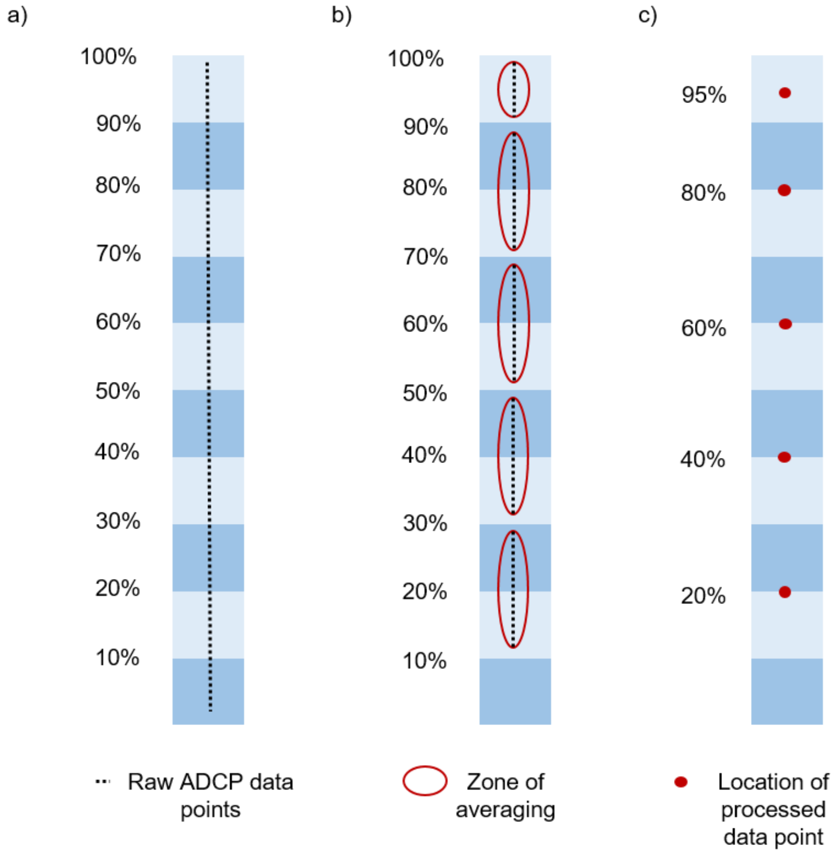


Figure 4.9 Diagram showing process of averaging the high-resolution velocity data (a) through the profile, the heights through which the averaging occurred (b) and the resultant 5 velocity points through the water column at 20%, 40%, 60%, 80% and 95% of the water depth (c).

The velocity and depth datasets were imported into Tecplot 360 and screened for error, as spurious depth data points can occur as a result of interference with the acoustic beams during the data collection (e.g. aquatic vegetation). These were easy to identify and mark for removal using an error velocity (v_d) value calculated by the equipment. v_d is calculated using the fourth acoustic beam in the Janus configuration. Three of the four acoustic beams emitted from the equipment measure v_e, v_n and v_z. The fourth beam also measures a vertical velocity vector. The v_d value is the absolute difference between two observed v_z vectors. Assuming the flow between the beams is homogenous, this difference should be zero to indicate no error (Teledyne RDI, 2011). The v_d can be used to estimate the degree of error of the velocity values in the dataset, but these values are not comparable between surveys due to variation in flows. Therefore, the $v_d$threshold used for screening differed depending on flow conditions.

The v_d threshold is visually defined in Tecplot 360 as the value at which retains the integrity of the dataset, the v_d values used for screening this dataset range between 0.15 and 0.2. Typically, values are higher for lower flows, possibly as a result of seasonal vegetation. Although this method of error identification is partially subjective, it removed extreme errors and was believed to be sufficiently accurate for this type of study. The depth data points identified as spurious and the v_d threshold were used to screen the data. The screened data were post-processed further using an Excel Macro to 'snap' the data points to grid nodes. This process calculated the mean value of data points within 1m of each grid node. Similar processes have been used by Dinehart and Burau (2005) and Tsubaki et al. (2012) to reduce noise and random error within ADCP datasets.

4.2.2 Data Interpolation

River monitoring data has been traditionally communicated through cross-sectional profiles, yet technological advancements have allowed for improved datasets and geospatial visualisation techniques. The use of Digital Elevation Models derived from the interpolation of bathymetric data is common a method to visualise bathymetry in fluvial geomorphology (Legleiter and Kyriakidis, 2008), and more recently, the geospatial representation of depth average velocity (Williams et al., 2015; Kriechbaumer et al., 2016). The spatial averaging process of the ADCP data reduced the density of the dataset. Kriging and IDW interpolation methods for ADCP data have been found effective in low density data (Tsubaki et al., 2012). Both Inverse Distance Weighting (IDW) and Kriging algorithms interpolate between data points using a distance weight. Both algorithms were applied in Tecplot 360 but the Kriging algorithm produced a smoother visual representation of the channel.

Anisotropic Kriging algorithms have been found to produce continuous geospatial representations of data with lower Root Mean Square Errors (RMSE) than ordinary Kriging algorithms (Merwade et al., 2006; Merwade, 2009). This approach has been applied successfully in the interpolation of bathymetric echo sounder data for use in hydraulic models (Merwade et al., 2005), thus, a similar approach was adopted in this study. Prior to interpolation, the grid and data points were converted to channel coordinates and compressed using an anisotropic compression factor (K). The channel coordinates were converted from British National Grid (Easting and Northing) to distance downstream along the channel centreline (S) and transverse distance from the channel centreline (T) (Fig. 4.11).

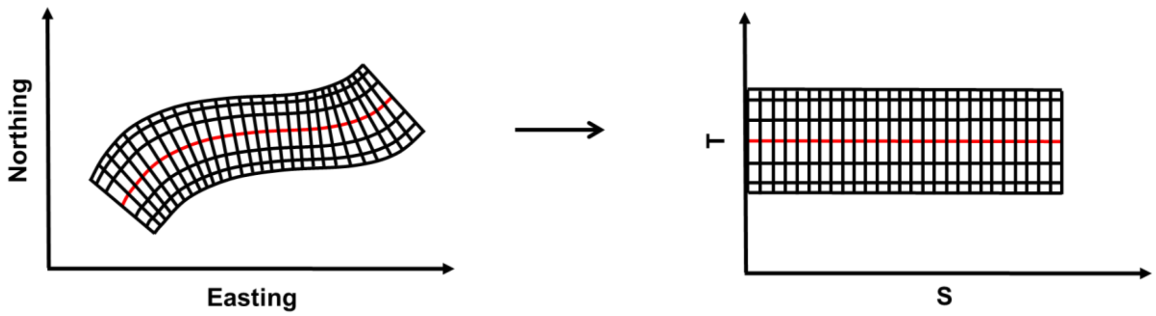


Figure 4.10 Conceptual diagram of the conversion between Cartesian (Easting and Northing) and channel (S and T) co-ordinates

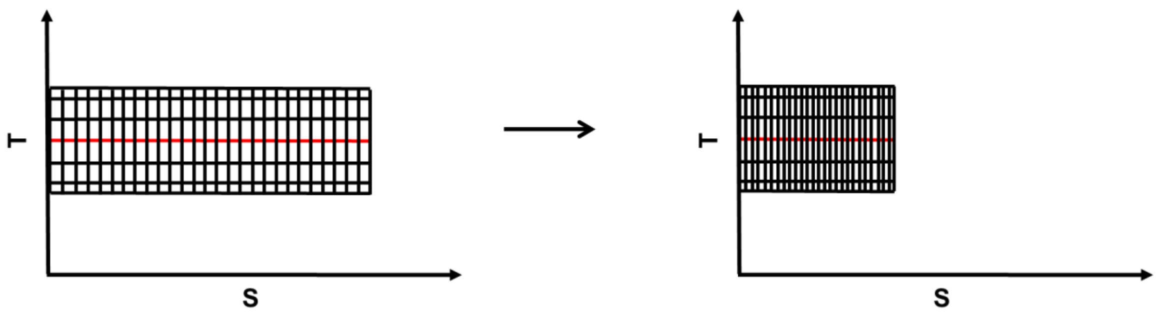


Figure 4.11 Conceptual diagram of the anisotropic compression value of K, from K=1 to K=2.

The anisotropic compression factor (K) also addressed issues associated with the paucity of data in the downstream, as the coordinates of the grid and data points are compressed by a given value of K along the centreline (Fig. 4.10). Therefore, during the interpolation process, the downstream is weighted by K over the cross-stream to reduce error. The appropriate value of K was evaluated between 2 and 15. Error between the pre-interpolation data points (p_1) and the interpolated grid (p_2) was analysed using a RMSE (Eq. 4.1) and a Mean Absolute Difference ($\bar{p}$) (Eq. 4.2) calculation, which were both computed in Matlab (Jamieson et al, 2013). When evaluated, all values of K between 2 and 10 had lower RMSE and $\bar{p}$ values than when K was not applied. K values greater than 10 saw the RMSE and $\bar{p}$ values elevated above those when K was not applied. The K value of 3 was found to have the lowest RMSE and $\bar{p}$ values for both the interpolation of elevation and velocity data at different surveys (high, moderate and low discharge), and was therefore applied hereafter for all interpolation routines of this study reach. It is anticipated that K could be site and survey specific (Merwade et al., 2006) and a similar testing procedure should be followed prior to application elsewhere.

(Eq. 4.1) $$RMSE = \sqrt{\frac{\sum(p_1 - p_2)^2}{n}}$$

(Eq. 4.2) $$\bar{p} = \sum\frac{|p_1 - p_2|}{n}$$

The Kriging interpolation routine performed in Tecplot used the following parameters Range (0.2), Zero (0.1), Drift (No Drift), Point Selection (Nearest N) and Number of Points (5000). The Range value defines the statistical significance of the data points for a given grid node between 0 and 1, with the latter indicating statistical significance of all data points for destined grid nodes. Range values between 0.2 and 0.5 are recommended (Amtec Engineering, 2001). The zero parameter defines the degree of smoothing with lower values preserving the data close to the target grid node. Given the data had already been smoothed, the lowest possible value to allow interpolation was used. A value of 5000 was chosen for the Number of Points parameter as it was the minimum value to allow the interpolation routine to run. A higher value was found to have a negligible impact on the results and significantly increased processing time. Larger values of the Range and Zero parameters were also tested to assess their impact on the resultant DEM and velocity planes by calculating RMSE and $\bar{p}$, but these were also found to have a negligible impact on the results.

The floodplain, bank, extent of wetted channel area, and in-channel elevation data (rectified depth data) comprised the input data into the interpolation routine for the DEM. Zeroed velocity floodplain and wetted channel area data were used alongside the processed ADCP velocity data during the interpolation to reduce the presence of historical features (see Section 3.2) occurring on the floodplain. Each of the 5 depth slices of velocity were interpolated separately, v_e, v_n and v_z were also interpolated separately before being resolved in 2D to produce a v_{Res} value which was used for data analysis (Section 4.3).

The limitations of the bank survey and a paucity of data created some issues in accurately representing the banks. The site is actively used for pasture which, until recently, had been unfenced and poaching of the banks by cattle has resulted in complex bank morphology throughout the reach (see Section 3.4). However, this is not fully represented in the topographic survey of the banks. The use of TLS or SFM for terrestrial surveys is recommended here for future river restoration monitoring surveys to improve the representation of bank morphology. Below the water surface, the channel can be assumed to be more accurately represented by the ADCP. However, above the water-surface level, the paucity of data led to some artefacts along the banks of the DEMs. Therefore, for contextual purposes, the elevation data above the water-surface surveys used interpolated elevation data from the 3 and 7 Months surveys. This was collected using the ADCP whilst these areas were submerged. For this reason, the analysis of geomorphic change was undertaken over the channel bed. However, velocity was analysed over the defined wetted area as velocity is measured independently of depth. Herein, the channel bed is defined as the wetted area surveyed during the as-built monitoring survey, as this is the area of maximum interest in the monitoring programme.

4.3 Data Analysis

The monitoring datasets were analysed in Matlab R2014a (Mathworks, 2014) using functions from the statistics toolbox, File Exchange (an online platform hosted by Mathworks for file sharing amongst the user community) and custom-built functions. Open source software would have been preferential to enhance the future applications of the research documented in this thesis. However, Matlab was used to extract the data from the equipment and was used for subsequent analyses to maintain consistency. Statistical analyses were performed using two spatial approaches (Fig. 4.12). Firstly, an evaluation of variables within the whole reach and over the riffle feature, which from herein shall be referred to as ‘Approach A’. The riffle feature was identified as the area which was directly modified during constructed, estimated as approximately 30m to 96m downstream of the upstream limit of the reach. Secondly, using a downstream moving window approach with a window width approximating average channel width of the dataset, which from herein shall be referred to as ‘Approach B’.

For geomorphic change analysis, average channel width was approximated as 30m. Conversely, for all other analyses (i.e. velocity, HMID and physical habitat simulations), average channel width was approximated as either 30m or 40m, for either low flows or moderate/high flows respectively. This Approach B was adopted following preliminary analysis, as it was observed that downstream trends were obscured when comparing the entire dataset with a single subset of the dataset. Approach A is still reported in Chapters 5 and 6 as it does provide some useful insights into the overall trends in the data. Four forms of analyses are undertaken in this research study to demonstrate the learning potential from river restoration, namely; geomorphic adjustment, velocity patterns, habitat heterogenity and physical habitat modelling.

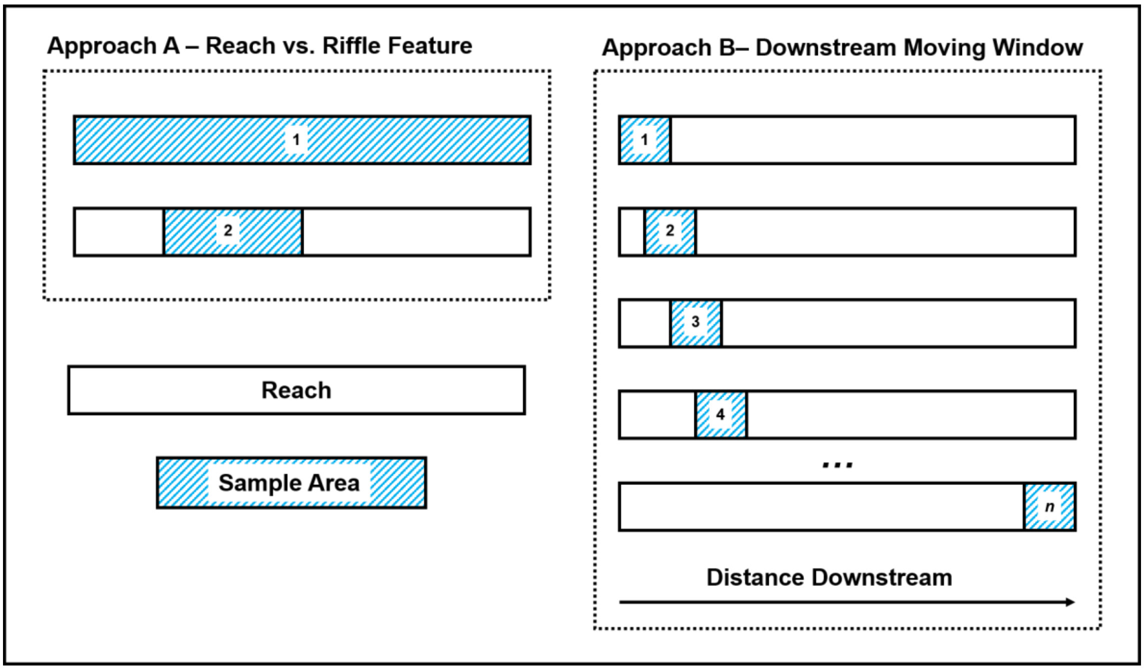


Figure 4.12 The two different approaches used to spatially analyse the RHES data set, Approach A – an overall and sub-reach assessment and Approach B – a moving-window approach.

4.3.1 Geomorphic Change

Geomorphic change was analysed using both Approach A and B. Summary statistics used in both approaches are summarised in Table 4.2, and include methods to describe the location, spread and shape of the distribution of data within the sample areas. As the dataset was interpolated onto a grid with irregular cell sizes, many of the summary statistics required weighting (w) to adjust for bias in cell size. Not all of these functions were standard functions of the Statistics toolbox, hence the use of the File Exchange and custom-built functions.

Table 4.2 A table of summary statistics adapted from Scown et al. (2015).

Weighted Statistic		**Description**
Mean ($\bar{x}_i$)	$\bar{x}_i = \frac{\sum_j x_j w_{ij}}{\sum_j w_{ij}}$	A measure of central tendency in the dataset.
Standard Deviation (s_i)	$s_i = \sqrt{\sum (x_j - \bar{x}_i)^2 w_{ij}}$	A measure of spread in the data set, that measures variability around the mean.
Coefficient of Variation (cv_i)	$cv_i = \frac{s_i}{\bar{x}_i}$	A measure of variability or dispersion of the data points relative to the mean.
Percentile (p_n)	$p_n = \frac{0.5\ (\sum\ w - w_n)}{2}$ Using the R-5 method.	Median = p_{50} Lower Quartile, = p_{25} Upper Quartile= p_{75} 2.5^{th} Percentile = $p_{2.5}$ 97.5^{th} Percentile = $p_{97.5}$
Quantile Coefficient Dispersion (Q_D)	$Q_D = \frac{p_{75} - p_{25}}{p_{75} + p_{25}}$	A measure of dispersion within the data set that is independent of the mean.
Percentile Coefficient Dispersion (P_D)	$P_D = \frac{p_{97.5} - p_{2.5}}{p_{97.5} + p_{2.5}}$	A measure of dispersion within 95% of the data set that is independent of the mean.
Skewness (sk_i)	$sk_i = \frac{\frac{1}{n}\sum_j (x_{j\,.}\ \bar{x}_i)^3}{\left(\sqrt{\frac{1}{n}\sum_j (x_{j\,.}\ \bar{x}_i)^2)}\right)^3}$	A measure of the shape of the data set distribution, a positive skew indicates bias from larger data points and a negative skew indicates bias from smaller data points.
Kurtosis (k_i)	$k_i = \frac{\frac{1}{n}\sum_j (x_{j\,.}\ \bar{x}_i)^4}{\left(\sqrt{\frac{1}{n}\sum_j (x_{j\,.}\ \bar{x}_i)^2)}\right)^2}$	A measure of the shape of the data set distribution, a higher kurtosis indicates a very peaked distribution dominated by a small range of values. A low kurtosis indicates a very smooth transition of larger range of values.
Rugosity (*R*)	$\frac{Area_{flat\ surface}}{Area_{actual\ surface}}$	A measure of surface roughness, a higher rugosity indicates a more complex surface. Calculated as a ration between the actual surface area of a cell (accounting for variability in height) and the flat surface area.

The datasets were non-parametric, therefore, the Kruskal-Wallis test was used in preference over a traditional analysis of variance (ANOVA) test to assess statistical significance. For all Kruskal-Wallis tests performed, a significance value (p) of 0.05 was used to test the validity of the null hypothesis that no significant change had occurred. If the p-value of less than 0.05 was returned, the alternative hypothesis was accepted. In both approaches to analysis, the Kruskal-Wallis test was used to identify whether changes in elevations in the distributions of the sample areas between surveys were statistically significant. In the moving window approach, the Kruskal-Wallis test was also used to identify whether changes in the distributions of elevations in adjacent sample areas of the channel were statistically significant.

In addition to these approaches, a DEM of Difference analysis was performed to evaluate the volumetric change (Z_{DOD}, Eq. 4.3) over the reach as a deviation from the baseline condition, as-built design, and as the change between successive surveys. Positive and negative Z_{DOD} values are associated with the addition (fill) and removal (scour) of material respectively. A value of 0, therefore, indicates no observed change. The Z_{DOD} value was contoured using a blue-white-red colour ramp (i.e. red = scour and blue = fill) in Tecplot 360 to provide a continuous 2D geospatial map of geomorphic change over the bed. The red-white-blue colour map has been suggested as a suitable diverging colour map for mapping environmental variables (Brewer et al., 1996) and has been used recently when using the DEM of Difference technique (e.g. Carley et al., 2012, Williams et al., 2015 and Curran et al., 2015).

Volumetric estimates V_{DOD} of geomorphic change are also calculated as the sum of the product of Z_{DOD} and the surface area (S.A) of the respective grid (Eq. 4.3). Variation in these volume estimates was assessed using the two spatial approaches to statistical analyses. This is a crude estimate due to large cell areas but provides an indication of the general trends of sediment transport within the reach, or sub-section of the reach, between two specified time periods. The magnitude of sediment transport processes between two periods in time can be estimated using the absolute volume change ($V_{|DOD|}$) (Eq.4.4). Furthermore, the V_{DOD} can be further analysed in terms of the volume of scour (V_{Scour}) and fill (V_{Fill}) (Eq. 4.5 and 4.6), to assess if the processes occurring within two specified time periods are promoting erosion or deposition. The samples of volumetric change were also identified as non-parametric, and therefore, a Kruskal-Wallis test was used for statistical significance testing.

(Eq. 4.3) $$V_{DOD} = \sum(Z_{DOD\,.}\ S.A)$$

(Eq. 4.4) $$V_{|DOD|} = \sum|Z_{DOD\,.}\ S.A|$$

(Eq. 4.5) $$V_{Scour} = \sum(Z_{DOD\,.}\ S.A) \quad if\ (Z_{DOD\,.}\,S.A) < 0$$

(Eq. 4.6) $$V_{Fill} = \sum(Z_{DOD\,.}\ S.A) \quad if\ (Z_{DOD\,.}\,S.A) > 0$$

Equipment error has been identified as a factor which could introduce a degree of uncertainty in the detection of morphological change, and minimum levels of detection (or thresholding) are used to distinguish real change from noise (Lane et al, 2003; Wheaton et al., 2010; Bangen et al., 2014, Bangen et al., 2016). The vertical equipment error over the bed is a function of ADCP depth error (1%), which was corrected to elevation (AOD) using water surface data using the RTK GPS error (± 0.015m). As morphological geomorphic is derived from two DEM data points, any geomorphic change estimate may have a maximum error in the region of (-0.015m - 0.5%D) ≤ Z_e ≤ (+0.015m + 0.5%D) and a minimum error in the region of (-0.015m +0.5%D) ≤ Z_e ≤ (+0.015m - 0.5%D).

The vertical error within the data presented in this study was evaluated using the deepest cross-section of the study reach (where error is theoretically maximised) for the surveys with the highest and lowest discharges. This analysis indicated that there were very narrow maximum vertical error bands of ± 0.0368m during low lows and ± 0.0560m during high flows (Fig. 4.13) due to equipment error. This is a worst-case scenario assessment, which conceivably could have implications for detecting very micro-scale geomorphological adjustments. The results of this study (Chapter 5) detected ecologically relevant micro-scale geomorphological adjustments, however, these were in shallower sections of the reach in which the band of error would have been narrower.

As the errors are likely to be distributed within a band of uncertainty, a Monte Carlo simulation was run on the elevation change values, to estimate the effect of equipment error on volume change estimates within the reach. The Monte Carlo simulation is a technique that uses random samples from a known population (range of error) to predict to a level of confidence, given as a probable range of error. Given the error of the ADCP elevation data (in this study) are a function of depth, the highest and lowest errors should be observed between the two highest and two lowest discharge surveys, respectively. The simulation was run on the volume change estimates between the lowest two surveys, the baseline (1.8 $m^3\ s^{-1}$) to 1 month (1.7 $m^3\ s^{-1}$) survey period, and the highest two surveys 3 Month (8.5 $m^3\ s^{-1}$) and 7 Month (13.1 $m^3\ s^{-1}$). The simulation was run at a 95%

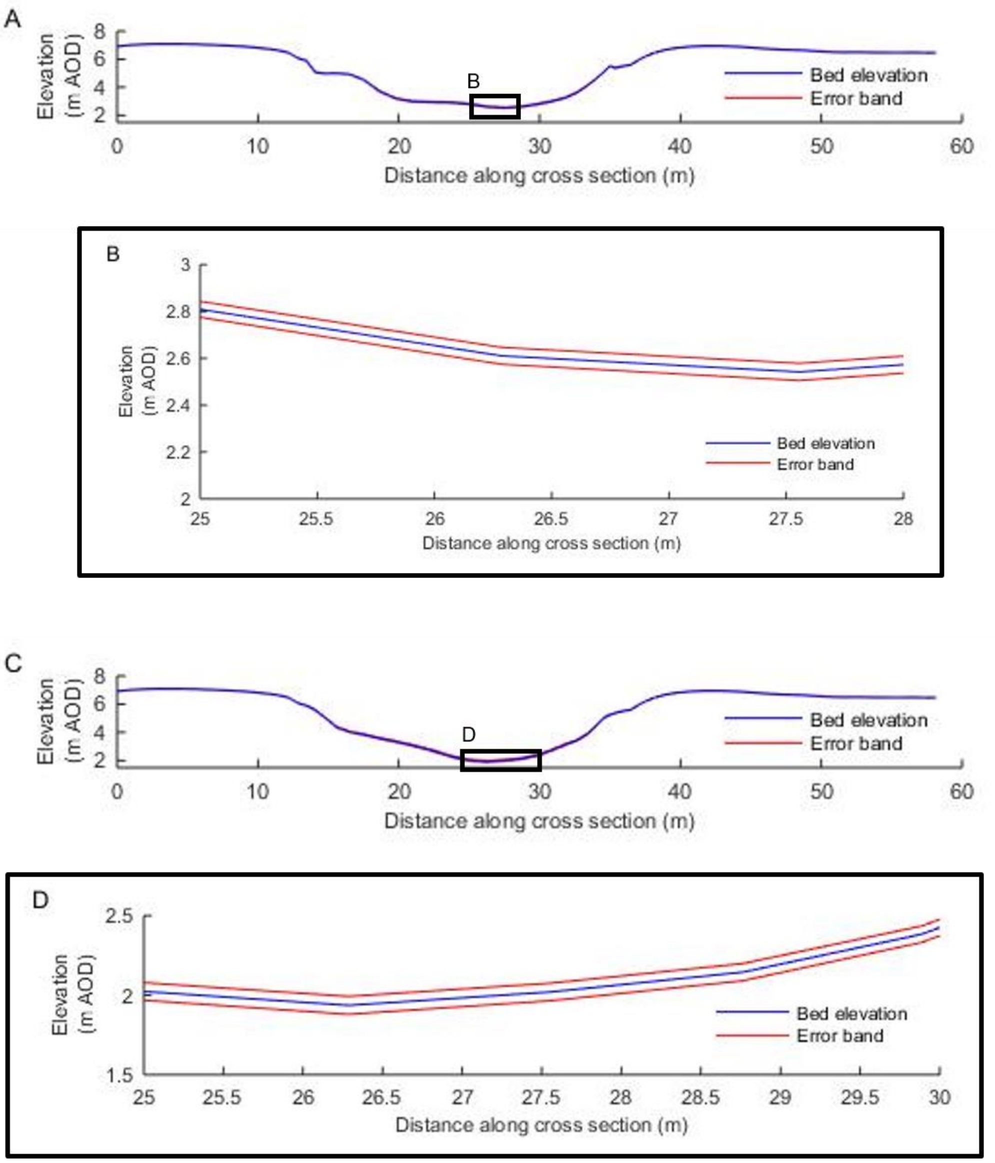


Figure 4.13 Potential impact of vertical equipment error on data reported within this study. A and B reflect the maximum error band in deepest section of the reach during a low flow survey and C and D reflect the maximum error band in the deepest section of the reach during a high flow survey. As the error band is so narrow, it is barely visually apparent in A and C.

confidence limit with 1000 iterations, the range of values produced as a result of the Monte Carlo simulation are reported in Table 4.3. The simulation results, as expected, indicated that the potential error was greater as discharge (and water depth) increased. However, the error relative to the magnitude of volume change was negligible. As the results suggest, equipment error does have the potential to influence volume change estimates, but in this study, the effect was likely very minimal and in all tested cases the error did not influence the volume change estimate value when rounding to one decimal place. This is a high degree of accuracy for this scale and application, in which an estimate to the nearest integer would probably be sufficient to draw meaningful conclusions. Consequently, further error modelling as documented in the literature was not felt necessary in this research study and further assessment of the accuracy of the equipment is beyond the scope of this study.

Table 4.3 Results of Monte Carlo simulation of equipment error on volume change estimates.

	Baseline to 1 Month			3 Month to 7 Month		
	Lower Estimate (m^3)	**Upper Estimate (m^3)**	**Potential Error Range (m^3)**	**Lower Estimate (m^3)**	**Upper Estimate (m^3)**	**Potential Error Range (m^3)**
Scour	59.63	59.67	0.04	92.53	92.59	0.06
Fill	345.64	345.74	0.1	459.50	459.75	0.25

4.3.2 Velocity Patterns

The depth average velocity patterns were also assessed using both Approach A and B. Statistical approaches described in the previous section (namely, summary statistics and Kruskal-Wallis testing) were also similarly applied, as preliminary analysis suggested the velocity dataset was also non-parametric.

4.3.3 Hydromorphic Diversity

Two methods were used in the assessment of habitat heterogeneity. Firstly, the calculation of the HMID (Eq. 4.7, Gostner et al., 2013) using Approaches A and B. Secondly, an agglomerative hierarchical clustering algorithm was applied to the velocity and depth data to identify statistically significant hydraulic patches within the reach. In Matlab, an agglomerative hierarchical clustering algorithm was applied as it grouped data to find natural breaks within the dataset, whereas other algorithms, such as K-means, required the user to specify the number of clusters as an input variable. Previous research using a hierarchical clustering method found ‘wards’ linkage function using the squared Euclidian distance to perform superiorly to other parameters when identifying hydraulic patches (Emery et al., 2003), and thus were adopted here. A height parameter was kept constant at the value of 30 for the clustering of each survey, this value produced an optimum number of clusters for further analysis.

(Eq. 4.7) $$HMID = (1 + CV_{Velocity}) \,.\, (1 + CV_{Depth})$$

More advanced clustering methods for the identification of hydraulic patches were available, such as fuzzy clustering (Wallis et al., 2012). However, these methods were considered excessive given the aims of the research project. Once identified, the perimeter of the hydraulic patches was evaluated using an alpha shape algorithm (1.5 search radius criteria). Subsequently, these patches were characterised using average velocity and depth values, and the dynamics of these patches within the riverscape were analysed using a series of configuration metrics (Table 4.4).

Table 4.4 Selected patch composition metrics used in the analysis of hydraulic patches and physical habitat quality (adapted from Frastats (McGarigal, 2014)).

Metric		Description
Simpsons Index of Diversity	$SID = \sum (n/_N)^2$	where n is the area of each patch and N is the total number of patches
Coverage	$P_{cov} = A_j$	where A_j is the total area of channel occupied by patch *j.* This metric can be used to evaluate the spatial abundance of patch *j.*
Patch Richness	$P_{Rich} = n$	where n is the number of different patches in the riverscape. This metric indicates the diversity of the riverscape.
Edge / Area Ratio	$P_{Comp} = \frac{A_j}{P_j}$	where P_j is the total perimeter of patch *j.* This measure can be used to estimate the fragmentation of patch *j* within the riverscape.
Fractal dimension	$P_{Frac} = \frac{2\ln(0.25\,.\,P_j)}{\ln(A_j)}$	This metric indicates the shape complexity of patch *j.*

4.3.4 Physical Habitat Simulation

Species suitability criteria can help evaluate whether geomorphological features have improved physical habitat conditions for species in the absence of ecological monitoring schemes. Physical habitat simulations using velocity and depth suitability criteria were performed in Tecplot 360. Suitability criteria for 3 fish species are evaluated in this research study, namely;

- Brown trout (Salmo Trutta), the target species of the RHES.
- Roach (Rutilus rutilus), a species documented in the reach prior to restoration; and,
- Dace (Leuciscus leuciscus), a species documented in the reach prior to restoration.

The suitability criteria ranged between 0 and 1, with the latter value indicating optimum conditions. Other more advanced physical habitat techniques such as fuzzy suitability models (e.g. CASiMIR (Mouton et al. 2007)) were also available. However, these methods were considered excessive given the aims of the research project. The velocity and depth suitability curves (Appendix A) used in this study are from a UK based-study by Armitage and Ladle (1991). These were seen as more transferable to the River Rother than those developed outside of the UK. The suitability criteria for each species is given at four life stages; spawning, fry, juvenile and adult, as species often require a range of habitats throughout their life cycle. A Habitat Suitability Index (HSI) was calculated for both velocity (V_{HSI}) and depth (D_{HSI}), the overall V_{HSI} was calculated as an average of the V_{HSI} of each velocity slice within the survey to account for the usable space within the water column. A Global Habitat Suitability Index ($GHSI$) was calculated using a geometric mean of the V_{HSI} and D_{HSI} (Eq. 4.8).

$$\text{(Eq. 4.8)} \qquad GHSI = \sqrt{V_{HSI} \, . D_{HSI}}$$

The $GHSI$ was mapped in Tecplot 360 to produce continuous spatial representations of Physical Habitat Quality. Subsequently in Matlab, the suitable area was categorised according to low ($GHSI$ < 0.4), moderate (0.4 ≤ $GHSI$ < 0.7) and high ($GHSI$ ≥ 0.7) habitat quality in a similar approach to Brown and Pasternack (2009). Using spatial approach B, downstream patterns in physical habitat quality were evaluated for each species and life stage. Taking the reach as a whole, the configuration of the low, moderate and high-quality habitat patches were assessed. Similarly to the habitat heterogeneity assessment procedure, an alpha shape (1.5m search radius) was used to identify the patches and calculate total suitable area. Similarly, configuration metrics used in the analysis of this data included a measure of patch fragmentation, richness and shape complexity (Table 4.3).

The physical habitat performance between surveys, based on both habitat heterogeneity and physical habitat simulations, is summarised using a subjective classification that considers all of the indicators calculated. For example, physical habitat performance using simulations would be classified using total suitable area, the quality of the suitable area and the fragmentation metrics. Relative to the performance of the previous comparable survey, the physical habitat performance is classified as either ‘significantly deteriorated’, ‘slightly deteriorated’, ‘no change’, ‘slightly improved’ or ‘significantly improved’. A more objective approach would be preferable but there does not appear to be the ecological evidence to support such a classification at present, this is a potential avenue for further research. Consequently, as a general rule of thumb, physical habitat performance is assumed to have improved where there has been an increase in total suitable area, an increase in the quality of the suitable area and a decrease in fragmentation as indicated by the metrics.

4.4 Summary

This chapter has outlined the monitoring programme of the RHES that was undertaken to capture the immediate geomorphological and physical habitat performance up to 18-months post-construction (January 2015). The design of the programme considered the principles outlined in Chapter 2 and is exploratory with respect to equipment and analytical methods used for river restoration monitoring. As with any monitoring schemes, there are limitations to this monitoring design such as a short period of baseline monitoring and lack of control site. These limitations may have been addressed if a longer period of pre-project monitoring was available. Chapters 5 and 6 present and discuss the results of the geomorphological and physical habitat performance of the RHES derived from this exploratory monitoring programme. Chapter 7 will discuss the value and practicality of undertaking this type of monitoring programme for routine river restoration monitoring.

5 River Rother Habitat Enhancement Scheme: Geomorphological Performance

5.1 Introduction

This chapter explores the results of the monitoring programme outlined in Chapter 4 to evaluate the geomorphological performance of the River Rother Habitat Enhancement Scheme (RHES). The aim of this chapter is to assess the resilience of the riffle feature over the monitoring period to prevailing catchment conditions, and the scheme's performance compared to similar forms and processes observed elsewhere (both natural and restored) that are documented in the literature. Based on the interpretation of the results, recommendations for future geomorphologically aligned management strategies are made. The insights gleaned here will inform a discussion in Chapter 7 in relation to physical habitat performance and potential physical habitat aligned management strategies for the River Rother.

5.2 Results: Morphological change over the bed

The DEMs of the study reach (Fig. 5.1) indicated an overall diversification in the morphology of the bed over the 18-month monitoring period. The DEM of the pre-restoration survey (July 2013) revealed the bed had a relatively uniform morphology, with an average elevation of 3.83 m above ordnance datum (m AOD) and low water surface slope of ~0.0001. The moving window analysis (Chapter 4.3) indicated little variation in mean elevation along the channel, except for an area of lower elevations in the last 30 m of the reach (Fig. 5.2). The lowest elevation within the reach (2.43 m AOD) was observed in this area, and henceforth this area shall be referred to as the 'downstream pool'. The channel form was largely uniform, as reflected by low Quantile Coefficient of Dispersion (QCD) of elevation (Fig. 5.2) and statistical insignificance between the distributions of elevations in most neighbouring areas of the channel (Fig. 5.3). Prior to restoration, only the downstream pool and the area where the head of the riffle was subsequently constructed exhibited a statistically different morphology from their neighbouring areas of the channel. The morphological variation of the latter was very subtle but appeared to have a mean elevation that was marginally higher than the neighbouring areas. Nonetheless, despite a general uniformity of channel form, the skewness, kurtosis and rugosity of elevations indicated there were some areas of mild bed surface complexity along the reach prior to restoration (Fig. 5.2, Appendix B). This was particularly evident in the downstream pool and over the riffle feature.

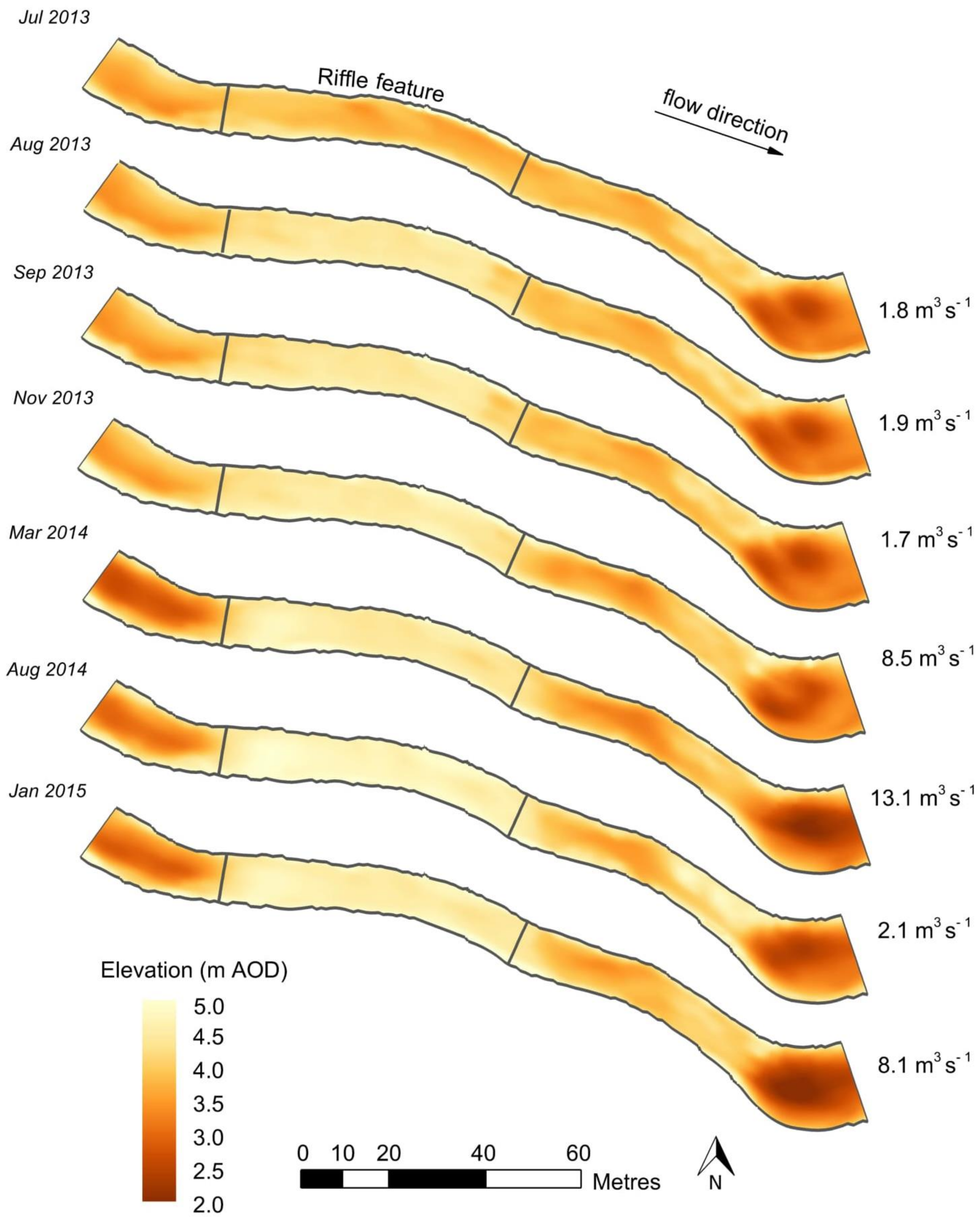


Figure 5.1 The DEMs of the 7 monitoring surveys between July 2013 (prior to restoration) and January 2015.

The as-built survey (August 2013) captured the morphology of the bed immediately post-construction (Fig. 5.1); approximately 337 m^3 of material was gained within the reach between the baseline and as-built surveys (Table 5.1). Given low flows were prevailing during the short time frame between the two surveys, the material gained within the reach was highly likely to be a result of the restoration activities. The riffle feature raised the profile of the bed by ~1 m, the form of the feature was constructed so that the elevation of the bed rose to a crest in the downstream direction (Fig. 5.4). Thus, the feature was implemented as designed. Some significant morphological change was observed outside of the riffle feature (Fig. 5.5), but this was restricted to the channel areas immediately upstream and downstream of the feature. Minimal change was observed in the downstream pool and it retained a largely similar form to the pre-restoration condition.

A downstream analysis indicated the distribution in elevations between neighbouring areas of the channel became more variable (Fig 5.3), suggesting that the diversity of form likely increased following restoration. This is supported by an overall increase in QCD for the reach elevations (Appendix B). However, when solely evaluated over the riffle feature the QCD decreased on the pre-restoration condition, signifying that the 'diversity' of form actually decreased over the riffle feature following restoration (Appendix B). Furthermore, rugosity values lowered along the reach, which suggested that the topographic surface complexity of the channel also declined following intervention. Skewness and kurtosis values peaked over the tail of the riffle feature (the point of highest in-channel elevation as constructed), which suggested the distribution of elevations were skewed by higher elevations at the channel margins (Fig. 5.3). When comparing the baseline and as-built datasets for this area of the channel bed, the 95% band of the elevation data was much higher and occupied a much narrower range of elevations than the baseline condition (Fig. 5.6). This indicates that the morphology of this area of the channel (when evaluated independently of the full feature and wider study reach) was more uniform and topographically smoother than the pre-restoration channel morphology. However, the wider reach revealed a greater diversity of bed elevations overall.

The 1-month post-construction DEM (September 2013) revealed little morphological change from the as-built DEM; during this period low flows also prevailed. Conversely, morphological changes from both the as-built and 1-month post construction DEMs were observed in the 3-month post-construction DEM (Fig. 5.1). Between these surveys, as expected during the autumn season, wetter weather contributed to a greater range of flows up to approximately a bankfull discharge. The maximum discharge recorded at Hardham Gauging Station was 15.8 m^3s^{-1}, however, this is a known high-in-bank flow and is also the maximum flow that can be gauged at this location. Therefore, the true maximum discharge within this period is unknown, but no significant out of bank flow events were reported during this time period, nor was evidence of such observed during subsequent site visits.

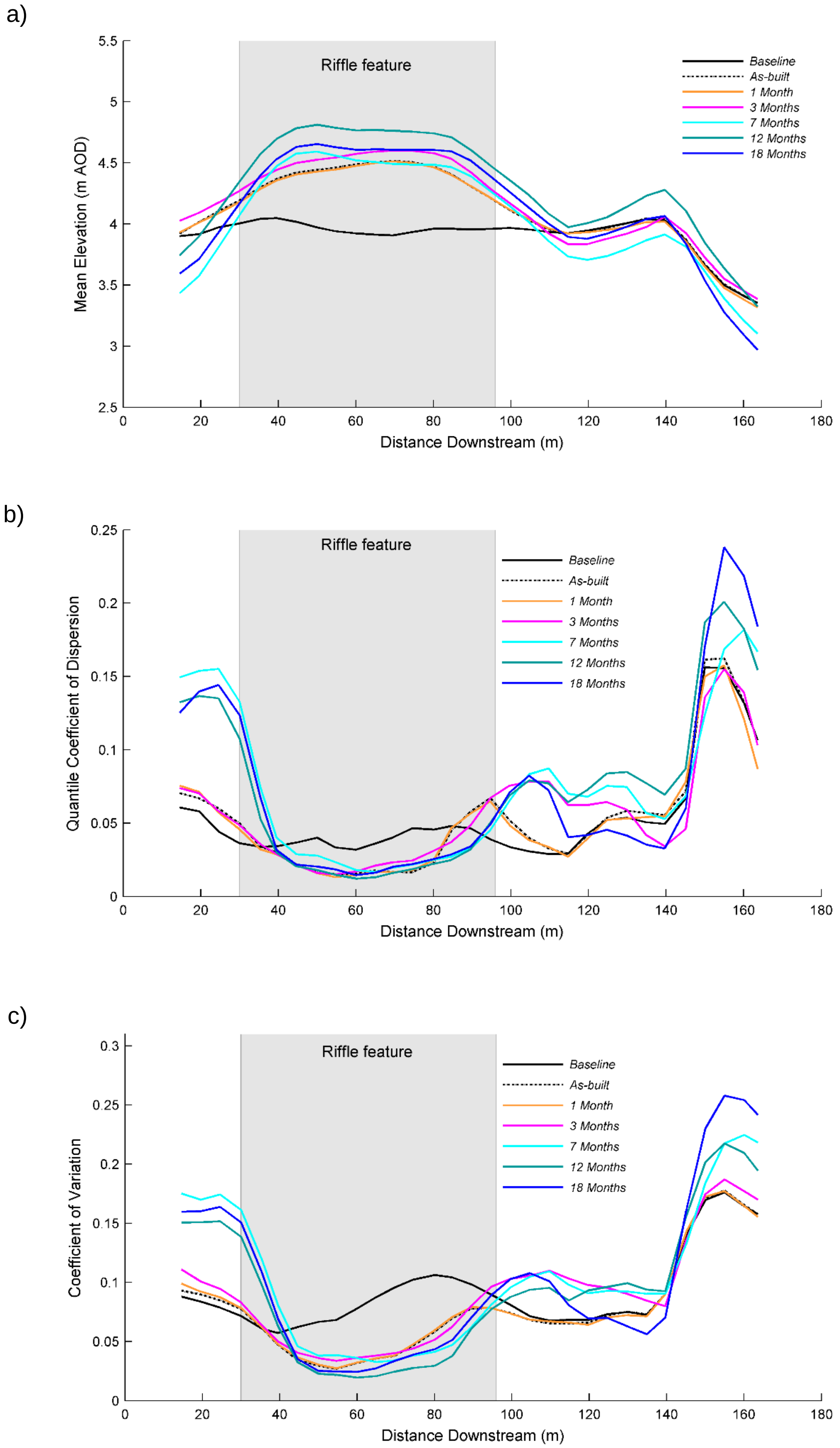


Figure 5.2 Moving window analysis of a) mean elevation, b) Quantile Coefficient of Dispersion, c) Coefficient of Variation, d) skewness, e) kurtosis and f) rugosity.

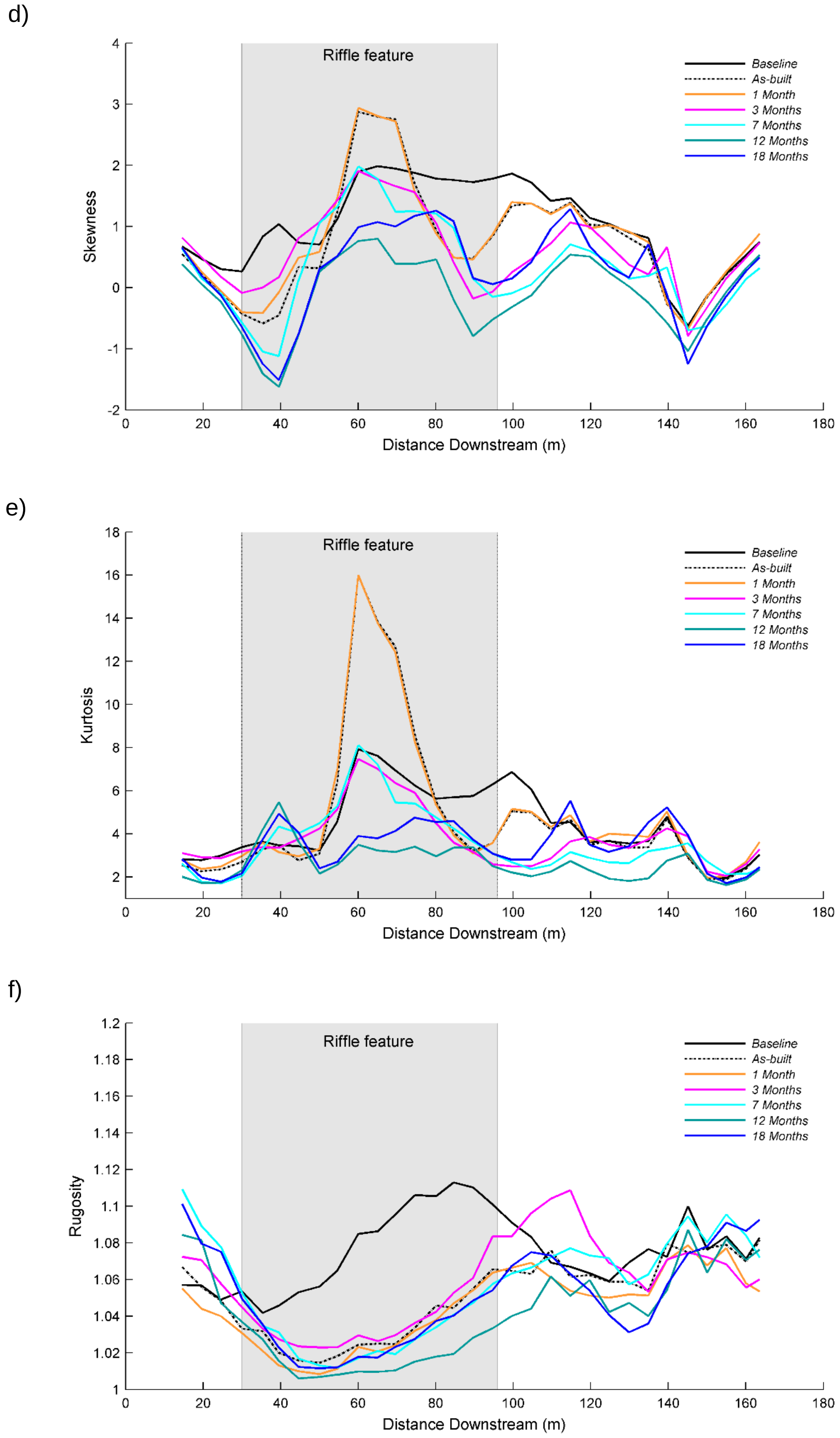
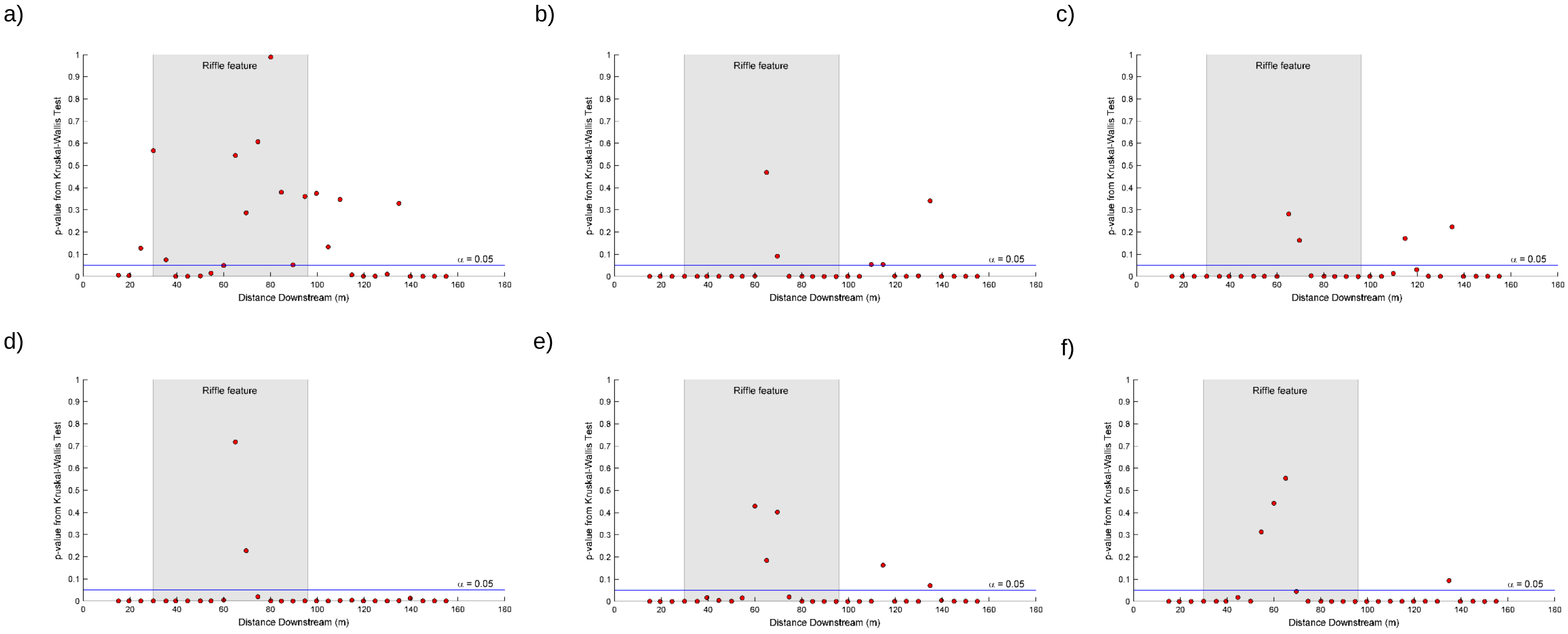


Figure 5.2 (continued) Moving window analysis of a) mean elevation, b) Quantile Coefficient of Dispersion, c) Coefficient of Variation, d) skewness, e) kurtosis and f) rugosity.



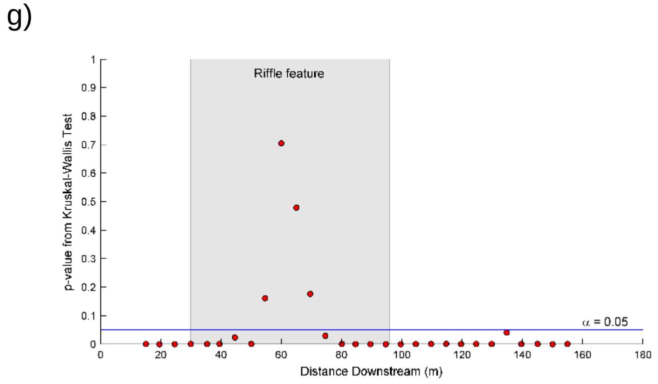


Figure 5.3 Moving window analysis using Kruskal Wallis statistical significance test of variability in the distribution of elevations along the channel a) pre-restoration, b) as-built, c) 1-month post-construction, d) 3-months post-construction, e) 7-months post-construction, f) 12-months post-construction and g) 18-months post-construction

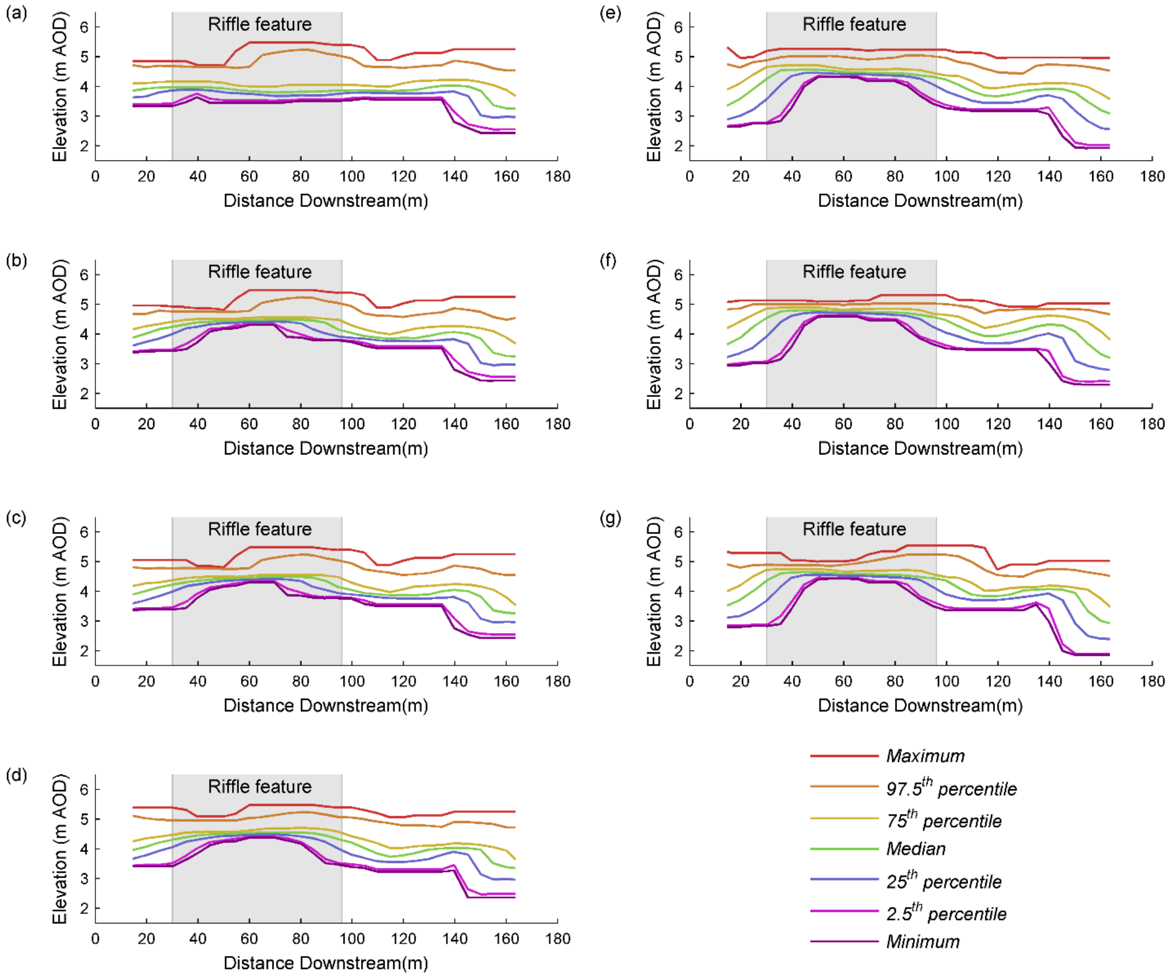


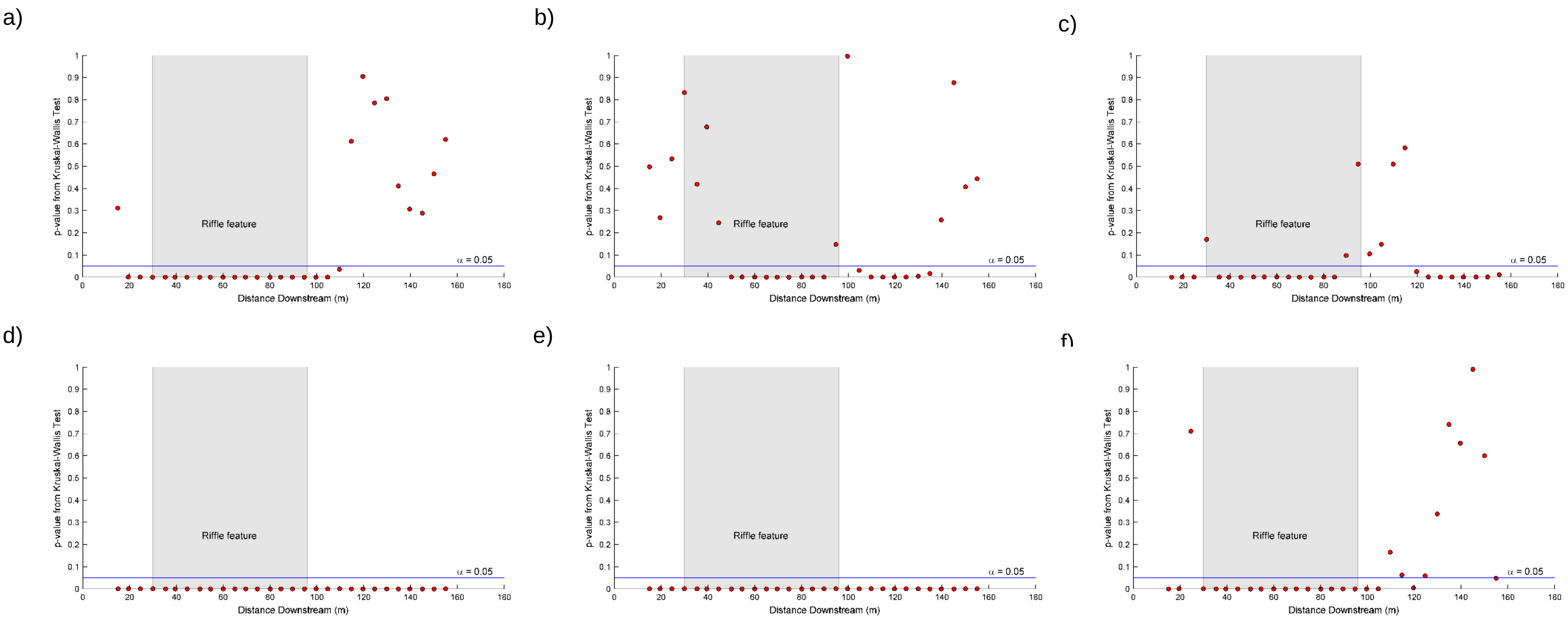
Figure 5.4 Moving window analysis of a range of percentiles of the elevation data from the a) pre-restoration, b) as-built, c) 1-month post-construction, d) 3-months post-construction, e) 7-months post-construction, f) 12-months post-construction and g) 18-months post-construction.

Some low levels of scour were observed immediately upstream and over the crest of the riffle feature, yet these morphological changes from the as-built survey were found to be statistically insignificant (Fig. 5.5). In contrast, deposition which dominated over the feature was found to contribute to statistically significant morphological changes within the reach. This deposition was reflected in an overall increase in mean elevation over the riffle feature by ~0.1 m. A net volume of ~86 m^3 of material was gained within the reach from the as-built condition; comprising ~95 m^3 of scour and ~181 m^3 of fill (Table 5.1). Mean elevation also increased over the wider reach but not to the same extent as observed over the riffle feature, as significant scour was observed downstream. Scour processes appeared dominant downstream of the feature, as the bed elevations over a ~50 m distance were lowered by ~0.5 m (Figs. 5.1 & 5.7). Through scour, the minimum elevation in the reach lowered to 2.35 m AOD.

During the first 3 months the riffle feature retained its structural integrity, despite some slight morphological change. This morphological change contributed to an overall increase in the QCD of elevations which suggests that the channel form throughout the full study reach had diversified compared to both the baseline and as-built conditions (Appendix B). However, minimal changes in the QCD of elevations over the riffle feature were observed from the as-built condition. This indicates that the riffle feature itself retained a simpler channel form than the pre-restoration bed morphology. Trends in the rugosity of elevations also suggested that the riffle feature was less topographically complex than the pre-restoration bed morphology (Fig. 5.2). Downstream of the riffle feature, however, where scour was observed the topographic complexity of the bed morphology increased. Minimal changes to form and topographic complexity were observed in the downstream pool.

Significant morphological changes to the bed were detected both over the riffle feature and the wider reach after 7-months post-construction in March 2014 (Figs. 5.3, 5.7, and 5.8). A greater range of flows would typically be expected during the winter season and the 2013/14 flood events resulted in the reach experiencing out of bank flows for an extended period prior to the 7-month post construction survey (Chapters 3 and 4). Total scour of ~ 460 m^3 was offset by ~93 m^3 of estimated fill, therefore, a net volume change of ~ -367 m^3 indicated substantial scour processes operated within the reach between 3 and 7-month post-construction (Table 5.1).

Scour of up to 1 m formed an area of lower elevations in the first 30 m of the reach and upstream of the feature, this area is henceforth referred to as the ‘upstream pool’. Similarly, the downstream pool also experienced excessive scour on the centre and left side of the channel, resulting in a new minimum elevation of 1.92 m AOD. Scour was not ubiquitous in the downstream pool, as deposition up to 1m was observed on the right side of the channel. Lower levels of scour (<0.5 m) were observed over the tail of the riffle feature and in the area ~10-25 m downstream of the feature. The deposition of material immediately downstream of the feature was not found to contribute to a significant morphological change from the 3-month post-construction survey (Fig. 5.5). Localised, yet significant, deposition up to 1m over the riffle head revealed an interesting reversal in the aspect of the riffle feature (Fig. 5.4 & 5.8). This would suggest that a riffle feature was retained following the flood events, but the structural integrity of the constructed feature was affected.



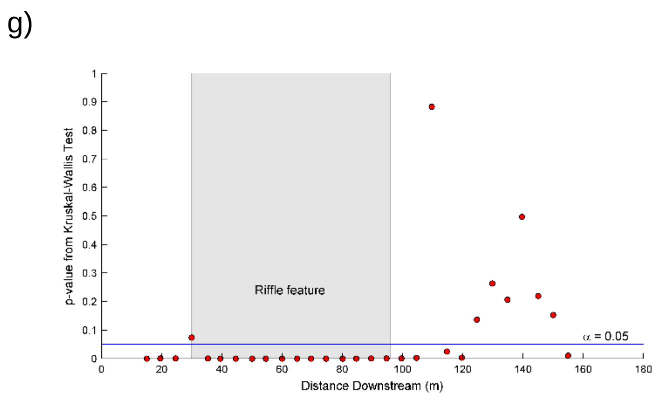


Figure 5.5 Moving window analysis using Kruskal Wallis statistical significance test of variability in the distribution of elevations between different surveys, namely, a) pre-restoration – as-built, b) 1 – 3-months post-construction, c) 3 – 7-months post-construction, d) 7 – 12-months post-construction, e) 12 – 18-months post-construction, f) pre-restoration – 18-months post-construction and g) as-built – 18-months post-construction.

	REACH VOLUME CHANGE ESTIMATES (m^3)				RIFFLE FEATURE VOLUME CHANGE ESTIMATES (m^3)			
	Net Change	Gross Change	Scour	Fill	Net Change	Gross Change	Scour	Fill
Sequential Surveys								
Baseline – As Built	309.0	365.4	-28.2	337.2	295.2	296.0	-0.4	295.6
As Built -1-Month	-23.0	107.5	-65.2	42.3	-7.4	24.4	-15.9	8.5
1-Month -3-Months	108.9	263.9	-77.5	186.4	65.1	67.4	-1.2	66.2
3-Months -7-Months	-367.1	552.2	-459.6	92.6	-7.6	81.7	-44.7	37.1
7-Months -12-Months	494.7	530.7	-18.0	512.7	149.9	154.9	-2.5	152.4
12-Months-18-Months	-363.1	427.7	-395.4	32.3	-92.1	102.6	-97.3	5.2
Change since baseline survey								
Baseline -1 Month	286.0	405.3	-59.6	345.7	287.8	292.7	-2.4	290.3
Baseline -3 Months	394.9	589.9	-97.5	492.4	352.9	353.1	-0.1	353.0
Baseline -7 Months	27.9	799.2	-385.7	413.5	345.3	357.7	-6.2	351.5
Baseline -12 Months	522.5	869.1	-173.3	695.8	495.2	503.8	-4.3	499.5
Baseline -18 Months	159.4	818.8	-329.7	489.1	403.0	415.8	-6.4	409.4
Change since as-built survey								
As Built -3 Months	85.9	275.8	-95.0	180.9	57.7	60.6	-1.4	59.1
As Built -7 Months	-281.2	535.7	-408.5	127.3	50.1	88.6	-19.3	69.3
As Built -12 Months	213.5	549.6	-168.1	381.6	199.9	208.5	-4.3	204.2
As Built -18 Months	-149.6	366.7	-169.4	197.4	107.8	128.5	-4.2	124.3
Other								
3 Months-18 Months	-235.5	508.1	-371.8	136.3	50.1	84.4	-17.1	67.3
7 Months -18 Months	131.6	300.9	-84.7	216.2	57.7	71.5	-6.9	64.6

Table 5.1 Volumetric changes of bed material over the full study reach and the riffle feature through the monitoring period. Gross change refers to total amount of change that has occurred within the given time period (i.e. both scour and fill), whereas net change refers to the residual of change that has occurred within the given time period.

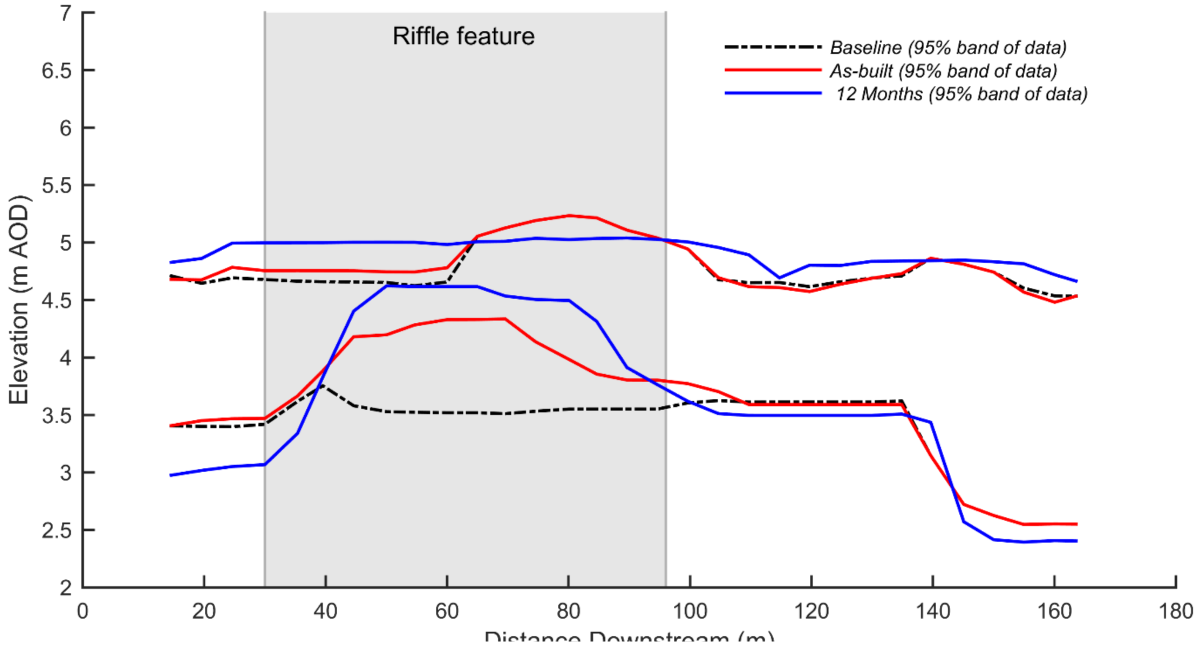


Figure 5.6 Moving window analysis of the 97.5th and 2.5th percentiles (95% band of the data) for the pre-restoration, as-built and 12-months post-construction surveys.

Overall, the reach exhibited a further diversification in the bed morphology on both the as-built and 3-month post-construction surveys, as reflected in overall increases in QCD of elevations. In the 7-month post-construction DEM, 3 distinct forms (2 pools and 1 riffle feature) were observed within the channel, as opposed to 2 forms following construction (1 pool and 1 riffle feature) and the 1 pool observed prior to restoration. When considering only the riffle feature, the QCD of elevation values increased on the previous post-construction surveys. This could suggest the feature had a more varied morphology following the floods than it did immediately following construction. Nonetheless, the QCD of elevations over the feature were not as high as that observed during the pre-restoration survey (Figs. 5.2 & 5.3). This indicates that the riffle feature (when considered alone) displayed a more uniform morphology than the baseline condition. Conversely, the QCD statistic indicated both the upstream and downstream pools exhibited a much more varied morphology than the pre-restoration condition. The rugosity of the bed elevations indicated a similar pattern, that the feature was also less topographically complex than the pre-restoration condition despite significant structural modification (Fig. 5.2). When only considering the riffle crests, the elevations over the head of the riffle feature following the 2013/14 flood events (new riffle crest) exhibited closer approximation to a normal distribution than the constructed riffle crest (old riffle crest ~ 50 downstream of new riffle crest). This would indicate that the new riffle crest had a more even distribution of elevations and potentially a less uniform morphology. However, over the remainder of the feature, elevations were positively skewed and leptokurtic, which would indicate an overall more uniform morphology.

The 12-month post-construction survey (August 2014) revealed further significant morphological change within the reach (Figs. 5.4, 5.7 and 5.8) had occurred over the 5 months following the notable flood events. Deposition appeared to be the dominant process within the reach over this period with a net volume change of ~495 m^3. This estimate comprised ~513 m^3 of fill and ~18 m^3 of scour (Table 5.1). This translated to blanket deposition of ~0.2 m over the reach, with some pockets of more extensive deposition (up to ~1 m) in the pools. Additionally, some small pockets of isolated scour were also observed. The mean elevation also reflected the trends of deposition along the reach, with the mean elevation increasing by 0.3 m over the whole reach, but only 0.2 m over the riffle feature (Appendix B). A net gain of ~214 m^3 of material was observed on the as-built condition, but the morphology of both the upstream and downstream pool features was generally unchanged from the previous survey. The morphology of the overall reach retained much of the diversity gained following the 2013/14 flood, as indicated by the QCD of elevations. The range of 95% of the data was more varied on both the pre-restoration and as-built condition of the channel (Fig. 5.6). Furthermore, the distribution of elevations within the reach exhibited the closest approximation to a normal distribution observed thus far. However, a further simplification of bed surface complexity was suggested by the rugosity of elevations throughout the reach (Appendix B).

Significant morphological change was observed between the 12-month and the 18-month (January 2015) post-construction DEMs (Fig. 5.4, 5.7 and 5.8). Scour of ~0.4 m was observed almost continuously along the reach, with some extensive scour of ~1 m in the downstream pool. The mean elevation of the reach decreased by ~0.2 m, but only 0.15 m over the riffle feature (Appendix B), this reflects the bias introduced from the extreme scour. An estimated net volume change of ~ -363 m^3 appeared to largely offset the extensive deposition observed following the 2013/14 floods (Table 5.1). However, the reach was in surplus of material by 132 m^3 from the 7-month post-construction survey. Despite this overall net gain of material following the 2013/14 floods, a net loss of ~150 m^3 of material was observed on the as-built condition and the morphology of both the upstream and downstream pool features was very similar to that observed during the 7-month survey. The overall diversity in channel form increased following the 2013/14 flood events, and the simplified bed surface complexity from the as-built condition were maintained (Figs. 5.2. and 5.3).

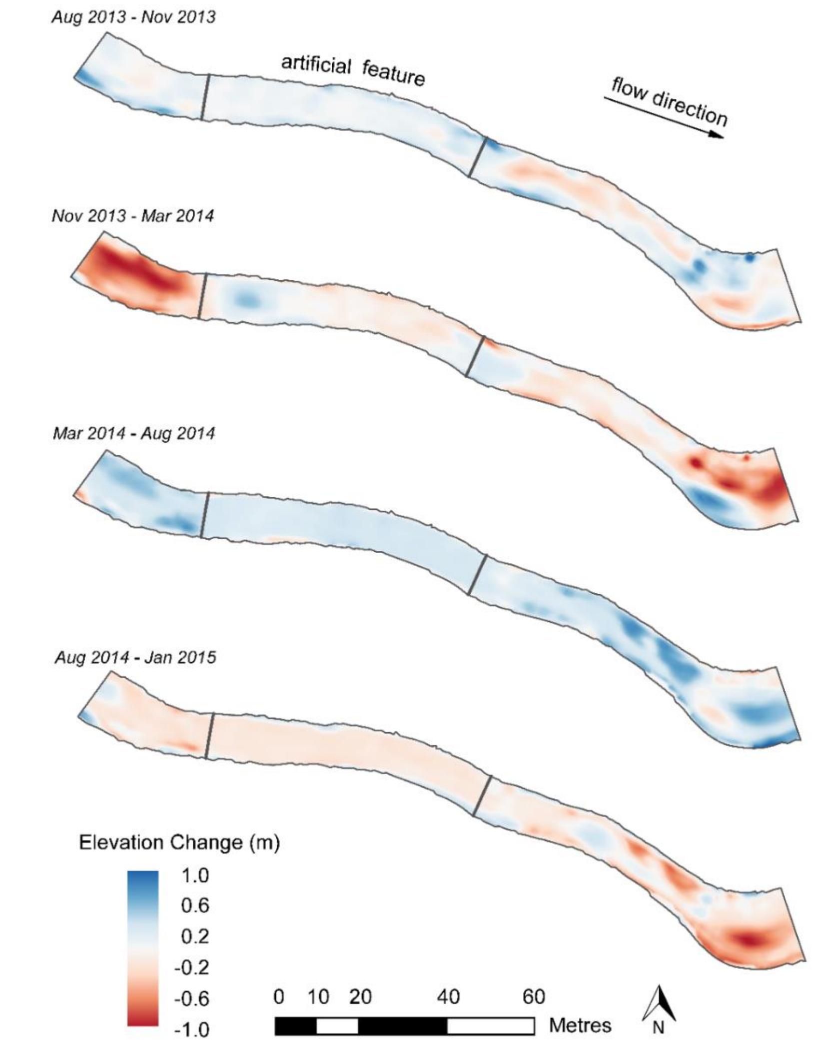


Figure 5.8 DEMs of Difference to highlight change between successive surveys

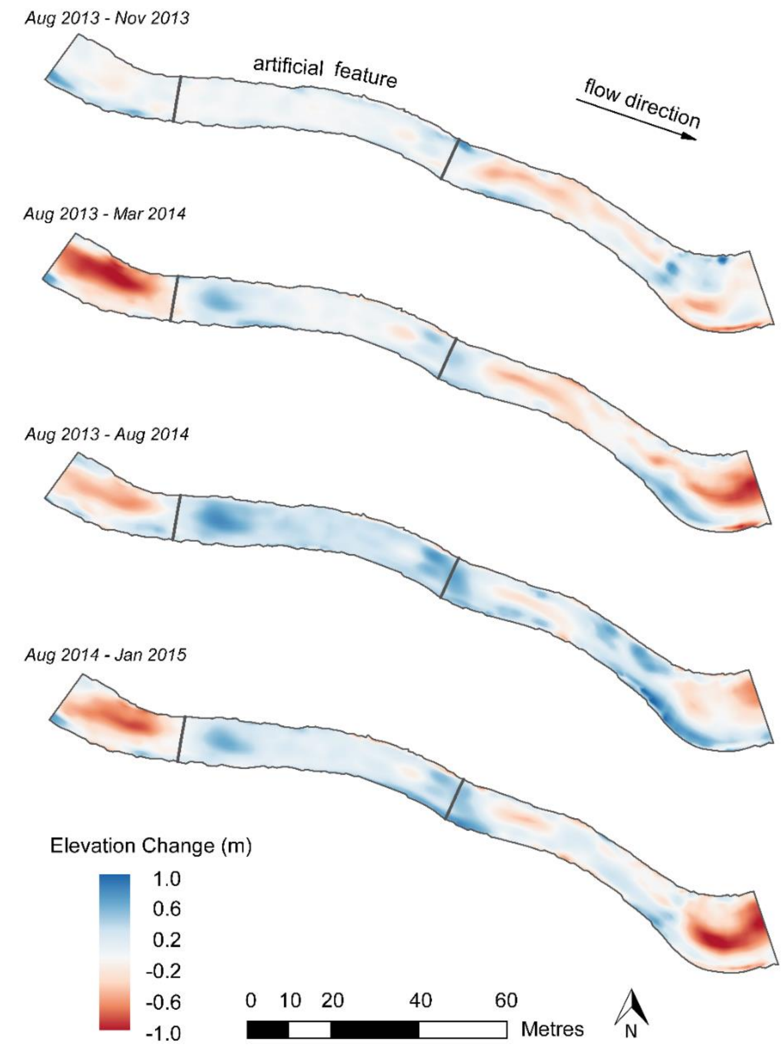


Figure 5.7 DEMs of Difference to highlight change from the as-built condition

5.3 Results: Velocity Patterns

Velocity was sampled in various locations throughout the water column using the ADCP as outlined in Chapter 4. However, to support the discussion on geomorphological processes, this section focuses on the results of the depth average velocity patterns (approximated here by the 40% depth level from the bed). The influence of the use of different velocity slices on the interpretation of physical habitat processes will be explored in Chapter 7. Significant variation in depth average velocity was observed within the reach between July 2013 (pre-restoration) and January 2015 (18-months post-construction). There were marked differences between the velocity patterns observed during the pre-restoration survey, and both the as-built and 1-month post-construction surveys (Fig. 5.9), all of which represent low flow conditions. Given the significant morphological change observed over the full study reach, modifications of velocity patterns were not unsurprising. The velocity pattern of the baseline survey (July 2013, 1.8 m^3s^{-1}) visually exhibited little of the diversity typical of sinuous rivers at low flows. The velocities within the entire reach ranged between 0.00 and 0.54 ms^{-1} and exhibited a mean velocity of 0.19 ms^{-1}. Higher velocities contributed to a slight positive skew of the distribution. The distribution was also platykurtic which suggested a reasonably uniform distribution of velocity within the reach (Appendix B).

The area of the channel in which the riffle feature was later constructed was observed to have a narrower range of velocities, between 0.00 and 0.43 ms^{-1}, and mean velocity of 0.22 ms^{-1}. The skewness statistic indicated that minimal bias was introduced by extreme values. However, the kurtosis statistic suggested the velocity distribution in this area was more platykurtic and thus highly uniform (Appendix B). Although the distribution within much of the reach was highly uniform, the distribution over the downstream pool was conversely peaky and positively skewed by an area of velocities elevated though localised channel narrowing upon entrance into the pool (Fig. 5.10). Statistical significance testing highlighted similarity in the velocity distributions along most of the reach but marked variability over the downstream pool as a result of this channel narrowing (Fig. 5.11).

Immediately post-construction (August 2013; 1.9 m^3s^{-1}) and 1-month post-construction (September 2013; 1.7 m^3s^{-1}) low flow velocities observed within the reach were considerably more varied than the pre-restoration condition, but were not significantly different from each other (Fig. 5.11). Both the as-built and 1-month post-construction surveys revealed higher values of velocity QCD over the reach than the pre-restoration condition (5.10). The velocities over the riffle feature showed a statistically significant variation from the pre-restoration patterns observed, whereas those outside the feature did not (Fig. 5.12). Therefore, this indicates that the diversification of velocities within the reach was a likely result of the construction of the riffle feature as part of the RHES.

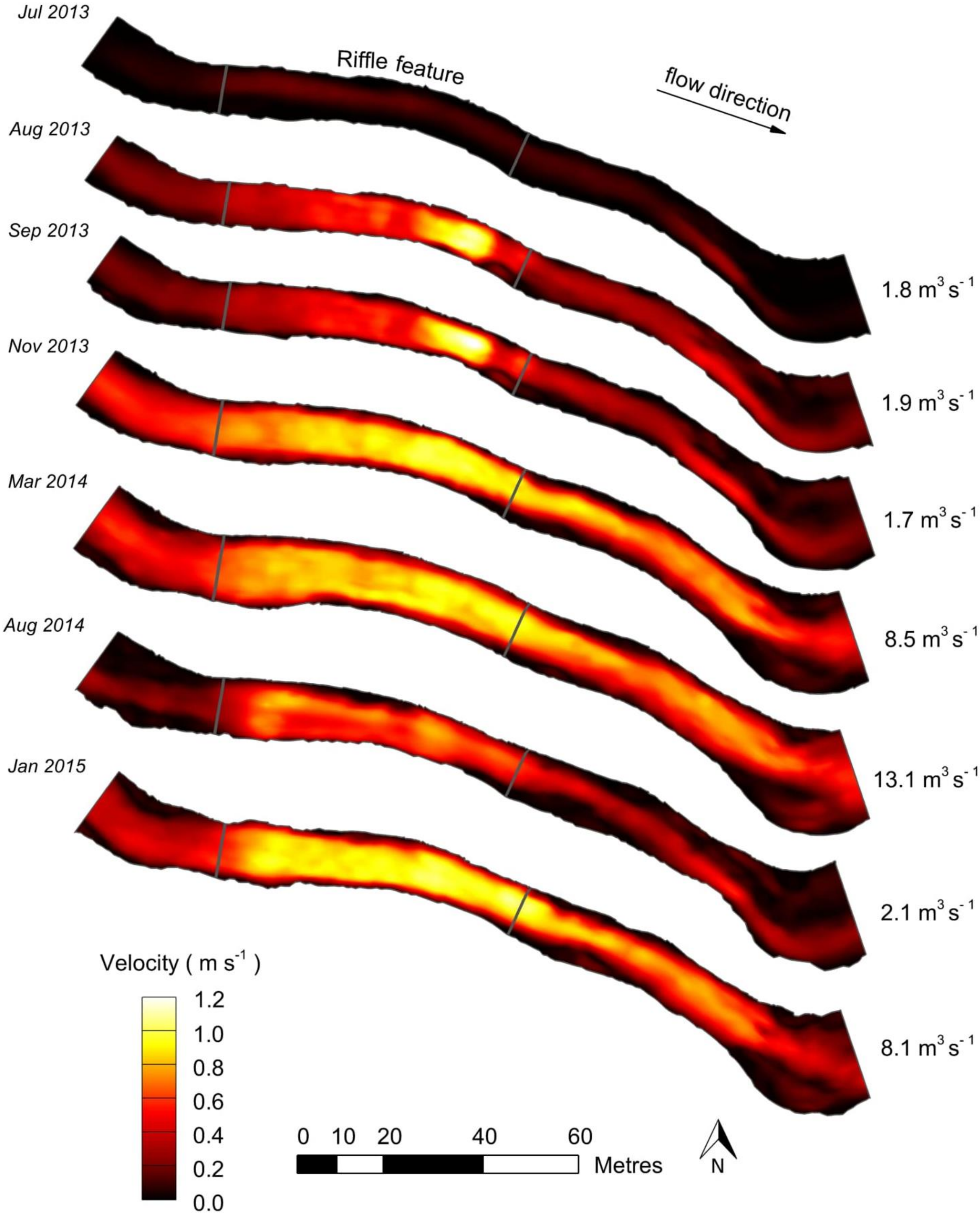


Figure 5.9 The depth average velocity slice (40% depth above the bed) of the 7 monitoring surveys between July 2013 and January 2015.

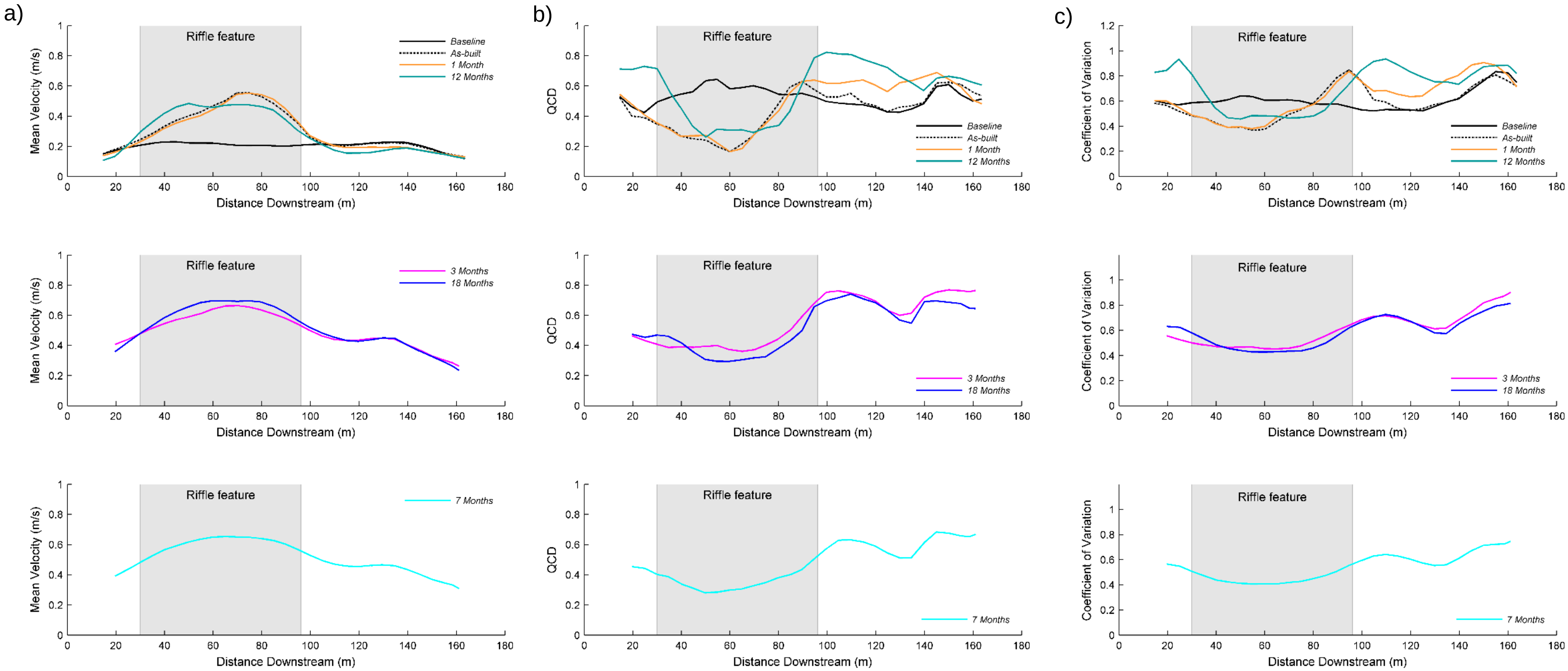


Figure 5.10 Moving window analysis of a) mean velocity, b) QCD of velocity, c) CoV of velocity, d) skewness of velocity and e) kurtosis of velocity (top = low flow, middle = moderate flows and bottom = high flows).

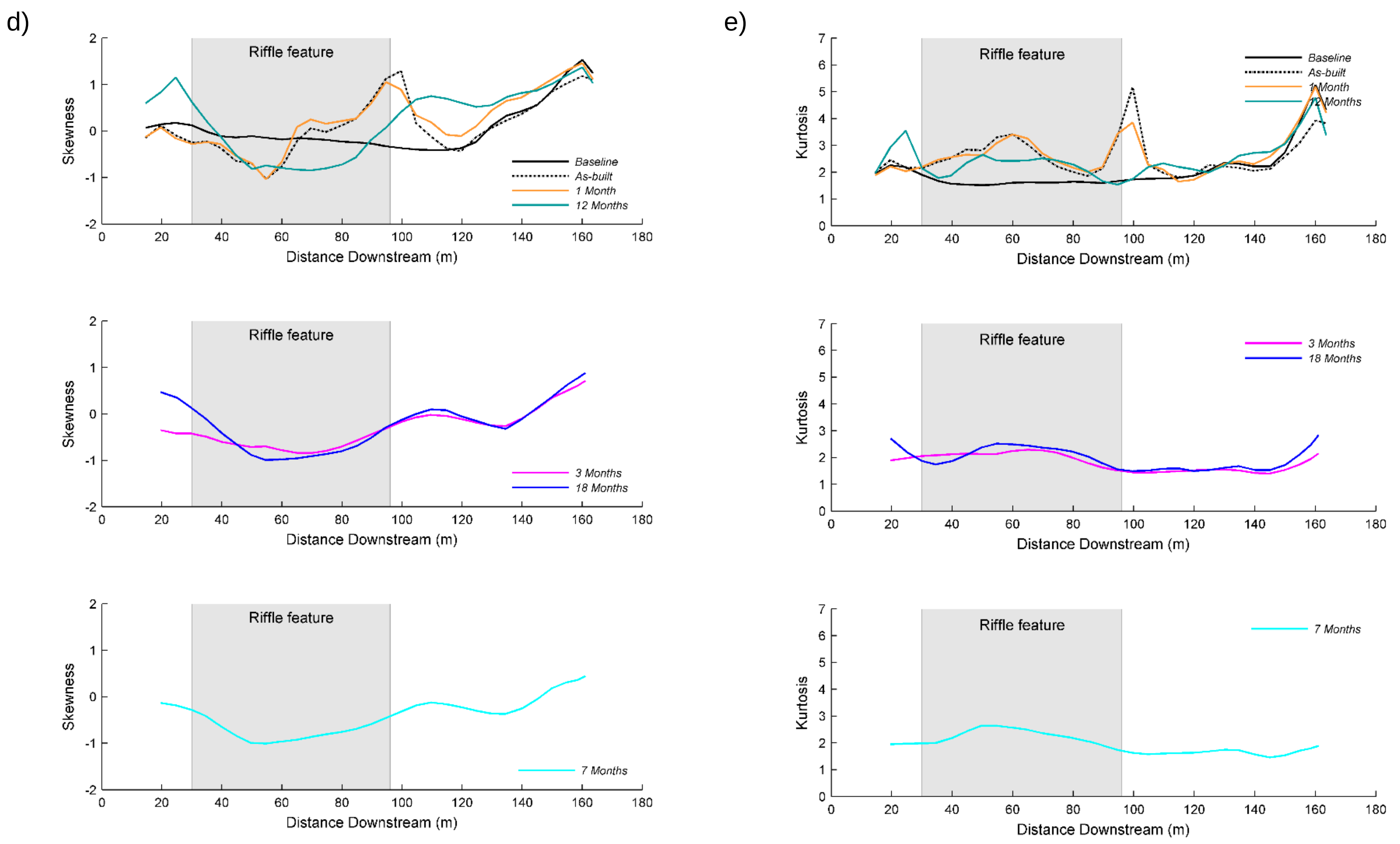
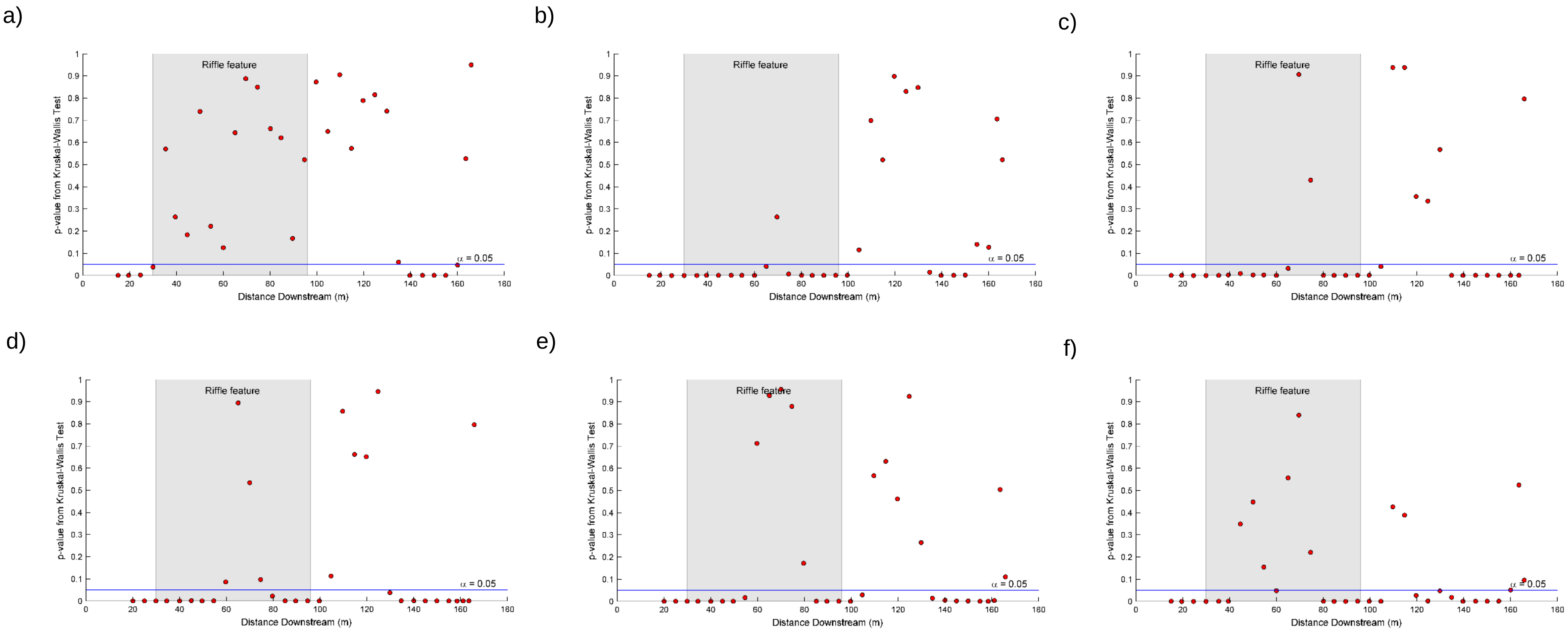


Figure 5.10 (continued) Moving window analysis of a) mean velocity, b) QCD of velocity, c) CoV of velocity, d) skewness of velocity and e) kurtosis of velocity (top = low flow, middle = moderate flows and bottom = high flows).



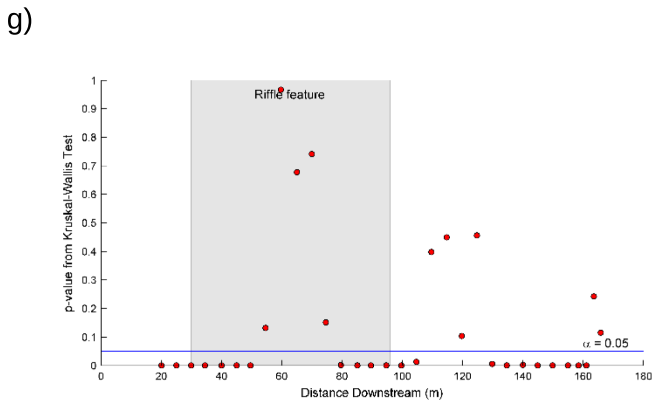


Figure 5.11 Moving window analysis using Kruskal Wallis statistical significance test of variability in the distribution of elevations along the channel a) pre-restoration, b) as-built, c) 1-month post-construction, d) 3-months post-construction, e) 7-months post-construction, f) 12-months post-construction and g) 18-months post-construction.

a)

b)

c)

d)

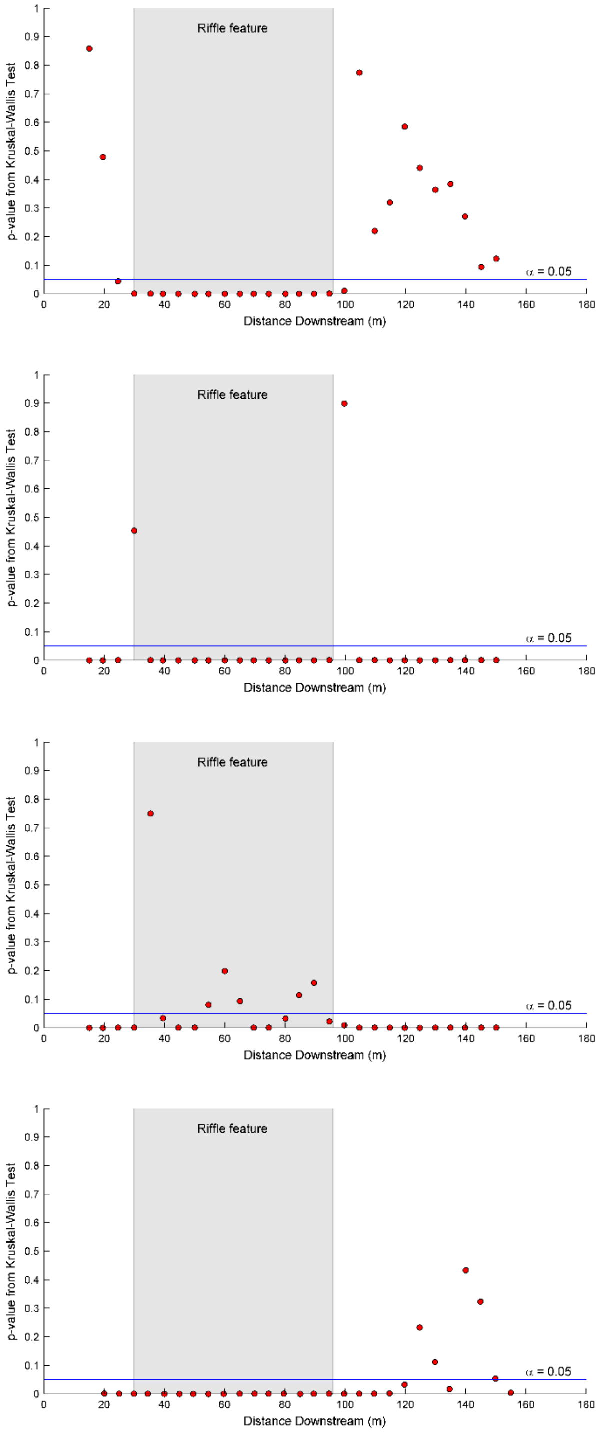


Figure 5.12 Moving window analysis using Kruskal Wallis statistical significance test of variability in the distribution of elevations between different surveys, namely, a) pre-restoration to as-built, b) pre-restoration to 12-months post-construction, c) as-built to 12-months post-construction and d) 3 to 18-months post-construction. Points which fall below the line y=α indicate a statistically significant difference between datasets, whereas points which fall above the line y=α indicates that the difference was not statistically significant.

The minimum velocities remained constant at 0.00 ms-1 in comparison with the pre-restoration condition, whilst the maximum velocities were elevated to 1.15 ms^{-1} and 1.24 ms^{-1} for the immediate and 1-month post-construction surveys, respectively. These maximum velocities were sited over the crest of the riffle feature (Fig. 5.13). The distribution of the velocities in the reach was leptokurtic and positively skewed in both the immediate and 1-month post-construction surveys. However, the velocities over the riffle feature alone were more normally distributed than the wider reach and exhibited lower QCD values than the baseline condition. This suggests a more coherent patch of similar, but higher, magnitude velocities.

The next comparable low flow survey was undertaken 12-months post-construction (August 2014; 2.1 m^3s^{-1}), and thus followed the 2013/14 flood events (Fig. 5.9). Significant modification of velocity patterns in the reach was observed, associated with the reported morphological changes since the as-built and 1-month post-construction surveys (see Section 5.2). Peak velocities within the reach observed over the crest (tail) of the feature following construction of the riffle in August 2013 were no longer present (Figs. 5.10 & 5.13). This was reflected by the transition of the velocity distribution from leptokurtic to platykurtic following the flood (Appendix B). Post-flood, the maximum velocity observed within the reach was sited over the new topographic high over the riffle head, however, this maximum velocity of 0.84ms^{-1} was significantly lower than the peak velocity observed immediately post-construction (Fig 5.13).

Statistical significance testing indicated that the velocity distribution observed after 12-months post construction was significantly different along the majority of the reach from the pre-restoration survey (Fig. 5.12). These results also indicate that the most significant changes within the reach from the as-built survey occurred beyond (both upstream and downstream) the riffle feature (Figure 5.14). In these areas, the velocity distribution became more positively skewed and increased in variability (Fig. 5.10). The velocity distribution over the riffle feature appeared to improve the coherence in velocities observed as reflected by the QCD of velocities (Fig. 5.10). Despite an overall lowering of peak velocities, the median velocity over the riffle feature increased (Fig. 5.13, Appendix B). Over the wider reach, the velocities were significantly more varied than those observed prior to restoration and immediately following construction.

Moderate and high flow baseline surveys were not conducted prior to restoration and so it was not possible to compare the 3, 7 and 18-month surveys (November 2013, March 2014, and January 2015) to pre-restoration conditions of similar flow magnitude. However, the 3 and 18-month surveys (Fig. 5.9) were undertaken either side of the winter 2013/14 flood events during flows of comparable magnitude (8.5 & 8.1 m^3s^{-1}, respectively). This presented an opportunity to compare the effects of flooding on the feature at a high within bank event. During the 3-month post-construction survey, velocities were still highest over the feature, with a maximum velocity of 1.01 ms^{-1}. However, whilst there appeared to be a significant variation in the median and lower quartile (Q1- 25th percentile) velocities along the reach, the upper quartile (Q3 – 75th percentile) and maximum velocities appeared to show less variation along the reach (Fig. 5.13). This suggests that the effect of the riffle feature (as constructed) in generating higher velocities (relative to the reach as a whole) reduced with increasing flow magnitude, i.e. the feature became drowned out. The coherence in velocity observed during low flows over the riffle feature and variability downstream of the feature were also observed during the 3-month post-construction survey, as reflected by the QCD of velocities (Fig. 5.10).

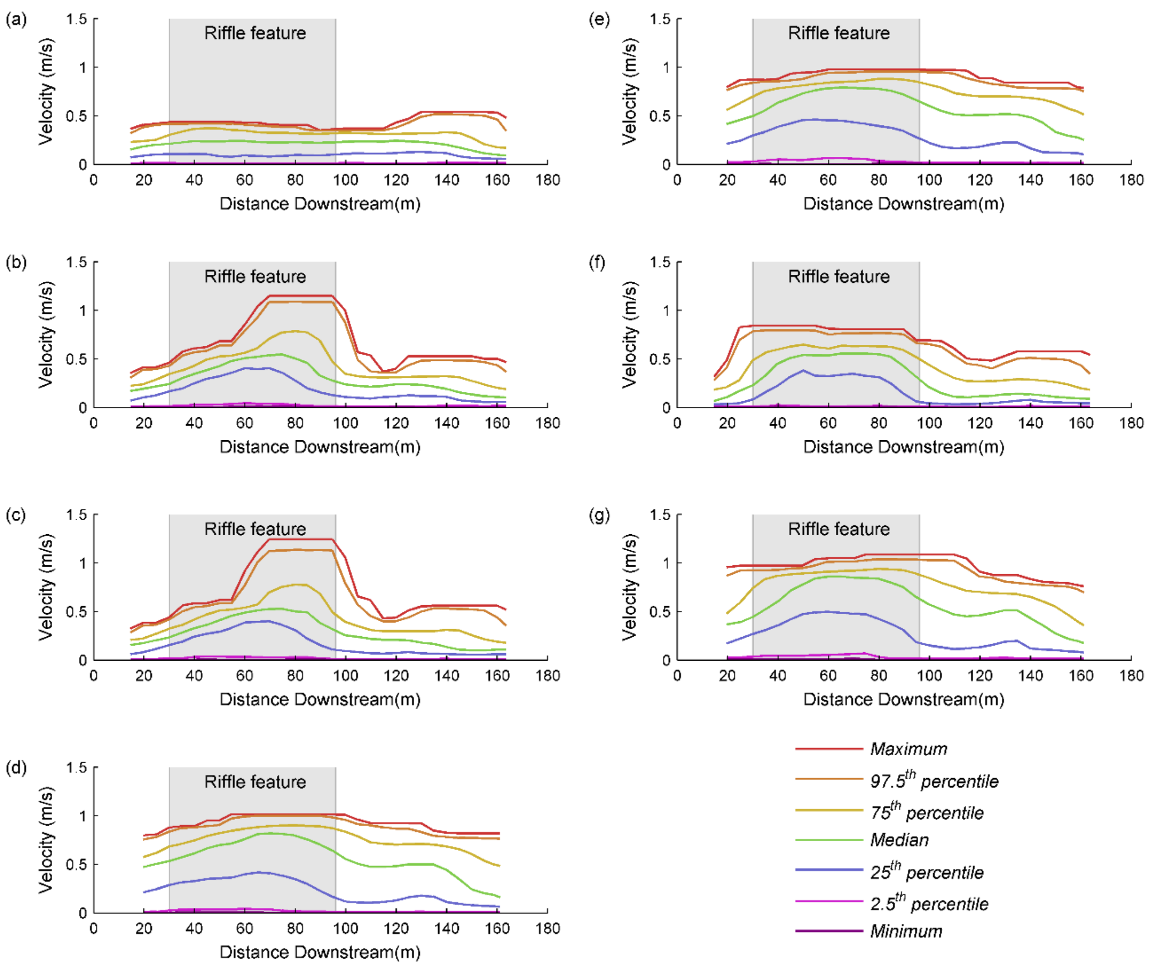


Figure 5.13 Moving window analysis of a range of percentiles of the elevation data from the a) pre-restoration, b) as-built, c) 1-month post-construction, d) 3-months post-construction, e) 7-months post-construction, f) 12-months post-construction and g) 18-months post-construction.

The velocity patterns observed in the 3 and 18-month post-construction surveys exhibited a similar pattern, such that the variation in velocity patterns over the downstream pool were not found to be statistically significant (Fig. 5.12). At low flows, the core of maximum velocity through the downstream pool was observed close to the right bank, whereas at moderate flows it was observed over the centre of the channel and closer to the left bank. Therefore, the shift in velocity patterns ties in with the patterns of scour and fill in the downstream pool observed during the monitoring survey (Section 5.2).

The maximum velocities showed little variation along the reach during both the 3 and 18-month post-construction surveys (Fig. 5.13). These results suggest that the riffle feature also became drowned out as discharge increased following the 2013/14 floods. However, the upper quartile velocity exhibited greater variability along the reach in the 18-month post-construction survey than in the 3-month post-construction survey. Furthermore, the median velocity over the riffle feature was observed to be higher during moderate flows following the 2013/14 floods, this is a similar trend which was observed during low flows (Appendix B). When considering the full study reach, these results suggest that feature was more influential on velocity patterns at moderate flows following the 2013/14 flood events. The lowest QCD of velocities within the full study reach were observed over the riffle feature in the 18-month post-construction survey as also observed in the 3-month post-construction survey. However, these QCD values were lower than observed from the 3-month post-construction survey (Fig. 5.10). These trends in the median and QCD velocities over the feature may suggest that the coherence of velocities improved over the riffle feature following the 2013/14 floods.

The 7-month survey was captured at a near bankfull discharge as the flood water receded and velocities were seen to exhibit similar characteristics to those observed during the 18-month post-construction survey (Fig. 5.9). Although there is no comparable pre-restoration high flow dataset for the 7-month survey, the results also suggest that the influence of the riffle feature on velocity patterns becomes drowned out as flow magnitude increases. This might indicate that the influence of the post-flood riffle feature on velocity patterns is most substantial below 8 m^3 (a moderate in-bank event), and further increases in flow magnitude have a minimal impact.

5.4 Interpretation of geomorphological performance

This section reviews the geomorphological results outlined in the previous two sections to discuss the performance of the RHES over the 18-month monitoring period in the context of the wider catchment (Chapter 3). This section discusses the results of the baseline monitoring results in the context of the dominant geomorphological processes at work prior to restoration. These findings provide a benchmark for the post-construction performance of the scheme. The marked range and succession of the flows experienced within the post-project monitoring period were quite serendipitous and presented an excellent opportunity to learn from the scheme's performance.

The succession of flows provided an excellent opportunity for two distinct discussions on the scheme's performance (Fig. 5.14). First, within three months following the construction of the riffle feature, it is assumed that a range of flows up to approximately a bankfull discharge were experienced within the reach. This affords the opportunity to discuss the geomorphological performance of the scheme with regards to a range of flows that is typical of a short-term monitoring project. Second, following this initial period, the occurrence of the 2013/14 flood events (approximate return period of a 1 in 50 year flow event) provided a rare opportunity to interpret the geomorphological performance of the riffle feature with respect to more extreme flow conditions.

Overall, however, an important observation made from the analyses of the monitoring data was that the interpretations of the scheme's performance had the potential to be influenced by the spatial scale over which the analyses are performed (e.g. the reach or sub-reach of interest). This is a key finding that has significant implications for the design of future river restoration monitoring schemes, which will be illustrated in this chapter and discussed again in Chapter 7. Therefore, when discussing the performance of the scheme within this section, the effect of spatial scale on the interpretation of performance and hence recommendations for adaptive management will be emphasised.

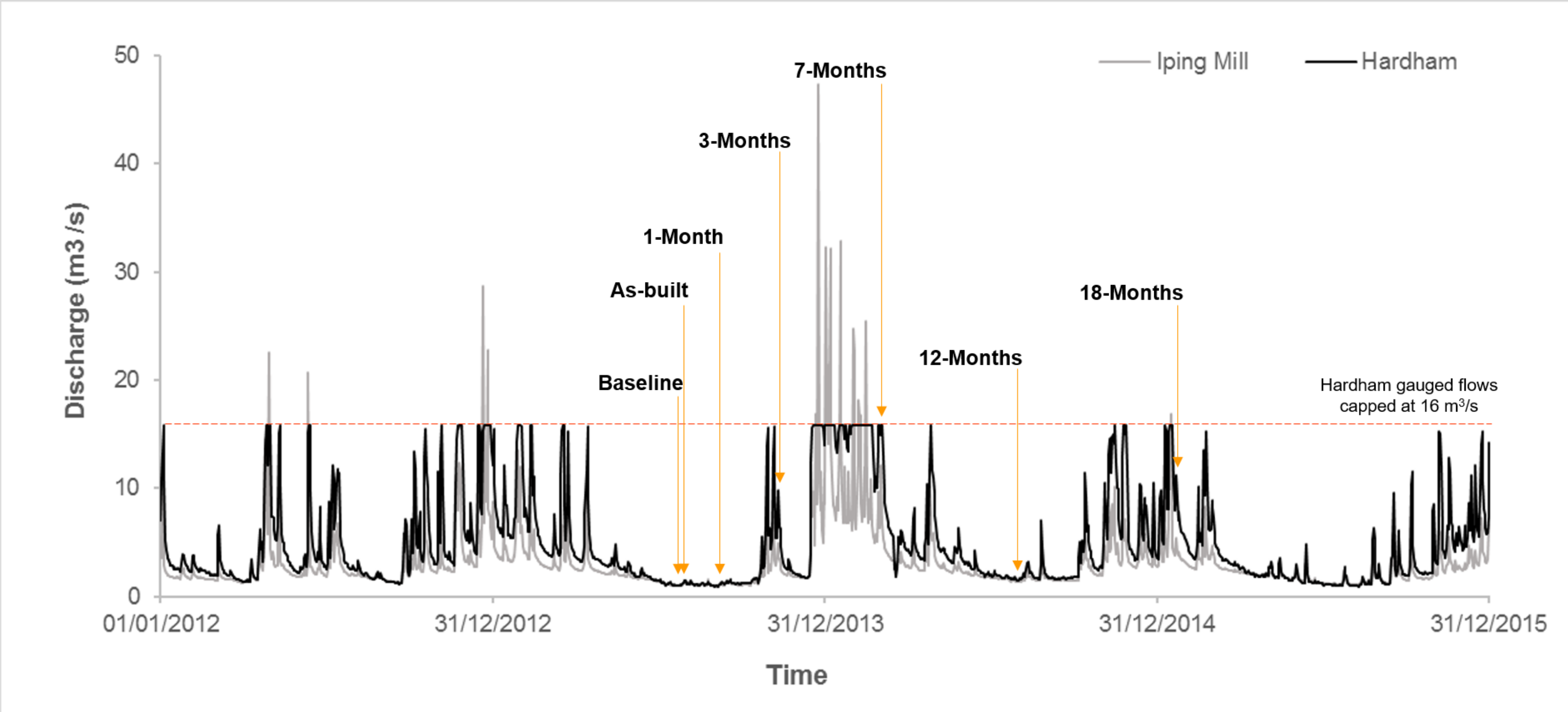


Figure 5.14 Daily flow data from the NFRA between the 1st January 2012 and 31st December 2015 for the River Rother at Hardham and Iping Mill gauging stations. The surveys undertaken as part of the RHES monitoring programme are highlighted to demonstrate the flows experienced within the reach during the monitoring period. Note that the flows at Hardham cannot be measured above 16 m^3s^{-1} but the flows at Iping Mill are indicative of the magnitude of events that occurred.

5.4.1 Pre-restoration geomorphology

Both the pre-restoration morphology and velocity were highly uniform throughout the reach with the exception of the downstream pool. This uniformity of the channel morphology at the reach scale is likely to have been the result of many catchment and local scale processes (Fig. 5.15). First, it is likely that the River Rother typically had a low capacity to transport sediment through the lower reaches of the catchment as a result of a naturally low water surface slope (see Chapter 3.2.4). It is also plausible that the capacity to transport sediment has also been limited by anthropogenic activities. For example, the water surface slope within the reach may have been artificially lowered by the impounding effect of Fittleworth Mill (~2.7 km downstream). As there has been a mill recorded in this location for at least 400 years (Vine 1995), this may have been a long-standing modification of the long profile and thus natural processes. The construction of other in-channel structures throughout the catchment has likely imposed a degree of longitudinal dis-connectivity to sediment transport. This may have limited the supply of gravel to the lower catchment, hence the dominance of sand in the bed material of the study reach.

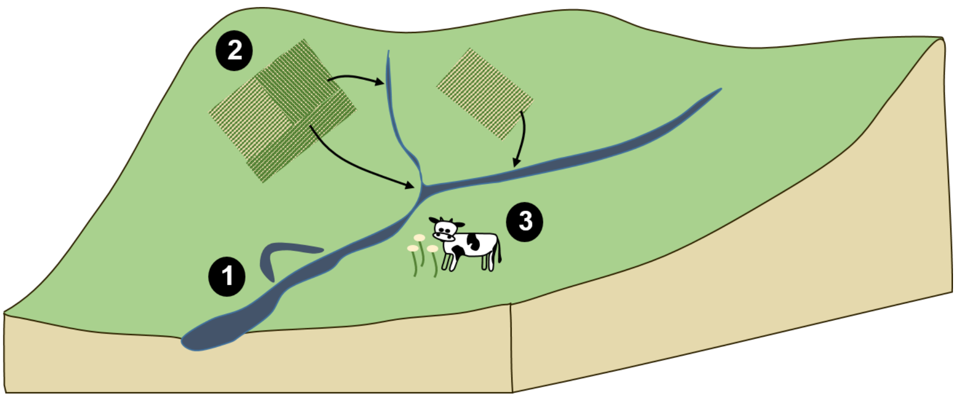


Figure 5.15 Conceptual diagram of catchment and local scale factors affecting the uniform pre-restoration morphology and velocity patterns. 1) Reduced capacity transport sediment (e.g. through channel modifications), 2) increased sediment supply (e.g. through direct sediment runoff from agricultural fields) and 3) shifting floodplain land uses (e.g. conversion of watermeadows to permanent pasture).

Furthermore, the construction of the Rother Navigation in the late 18th century is believed to have involved a significant enlargement of the channel geometry (widening and deepening) in the lower catchment downstream of Midhurst (Cox and Soar, 2017). This may have further reduced the capacity for sediment transport and increased in-channel sediment retention through reducing the opportunity for out of bank flows (Sear, 1996).

During the direct channel modification of Wallop Brook, Hampshire, during the 1980s, the enlargement of channel geometry also reduced the capacity for sediment transport. Immediately following modification, fine sediment accumulated in pools more significantly than in riffles which led to a homogenisation of bed morphology (Brookes, 1986; 1987). Therefore, it is possible that historical alterations to sediment transport processes in the River Rother have contributed to the lack of morphological diversity observed within the reach prior to restoration.

Second, recent studies within the catchment suggest that the delivery of fine sediment to the channel has increased in recent years as a result of the intensification of agricultural practices (Sear, 1996; Boardman et al., 2009). As discussed in Chapter 3, this may be due to a combination of catchment-wide sources (e.g. increased soil erosion risk from the intensification of arable agriculture) and also more local sources (e.g. bank erosion via floodplain land management and invasive species). Studies elsewhere have noted the accumulation of fine sediment in pools when sediment supply increases, particularly at low flows (Lisle and Hilton, 1992; Rathburn and Wohl, 2001). Thus, this could suggest that the increased sediment load to the River Rother may have also contributed to the uniform morphology observed within the reach prior to restoration through the infilling of pools. As the reach was only observed at low flows prior to restoration when pool infilling processes are most extensive, it is conceivable that the bed may have exhibited a greater morphological variability if observed during moderate or high flows. Therefore, this highlights the importance of baseline monitoring at a range of flows prior to restoration monitoring to fully understand the geomorphological processes operating within the impact area.

Third, cross-sections from the pre-restoration DEM suggest a degree of lateral dis-connectivity with the floodplain (Fig. 5.16). This might have promoted the retention of fine sediment within the channel through reducing the frequency of floodplain inundation and therefore, increasing the potential for reducing morphological diversity through infilling processes. Sediment cores from the floodplain in the local area have suggested frequent historical floodplain inundation (Walling and He, 1998) but a decrease in floodplain sedimentation rate was noted since the 1960s (Walling, 1999). These observations may suggest that the lateral dis-connectivity observed in the reach either intensified or post-dated the Rother Navigation.

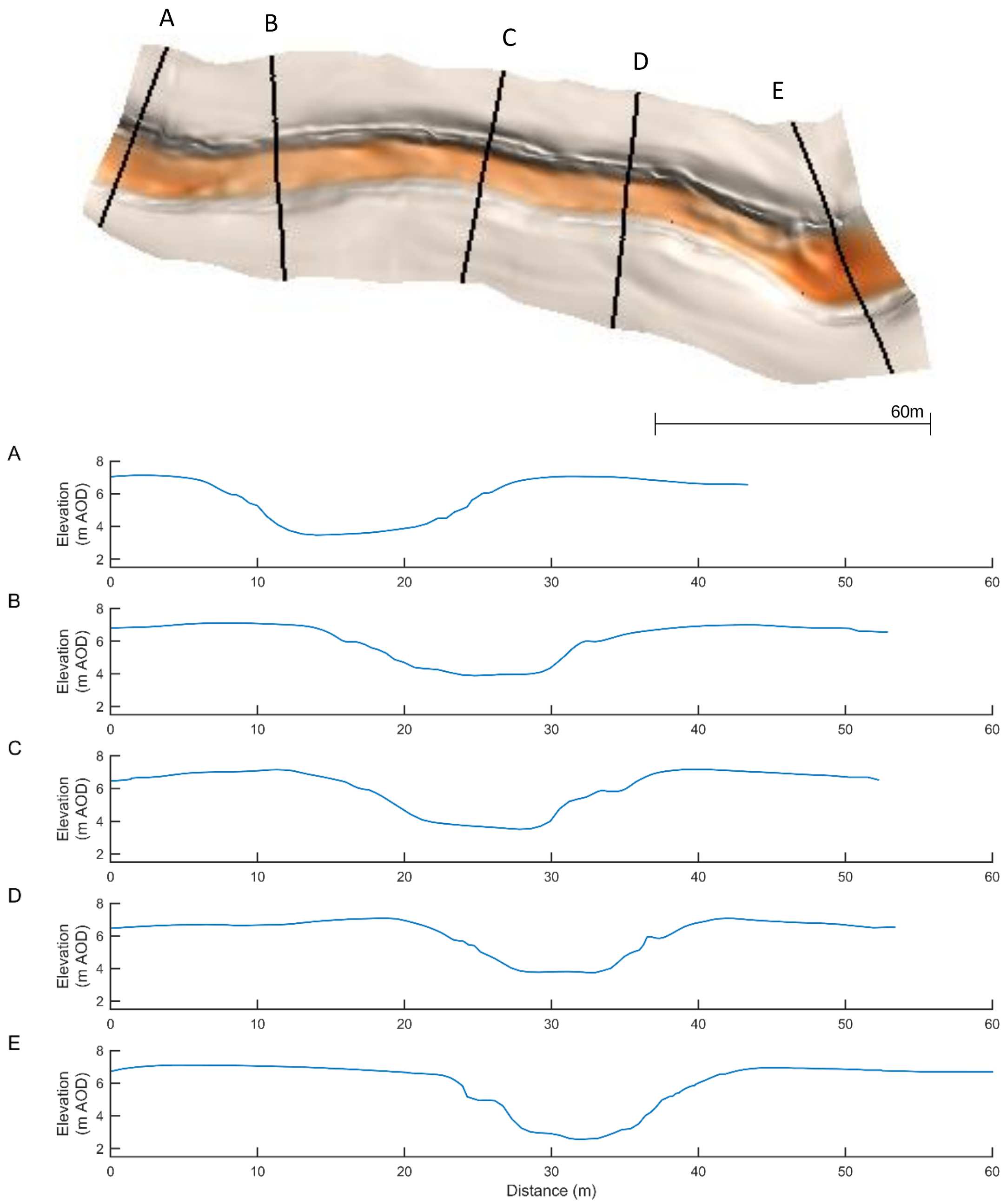


Figure 5.16 Cross-sections (looking upstream) of the channel and floodplain prior to restoration (July 2013).

The observations from the sediment cores could possibly reflect the impact of the changing nature of floodplain land use. Managed watermeadows were a common feature along much of the lower Rother floodplain for many centuries prior to the Rother Navigation and archival evidence suggests their persistence well into the mid-19th century (Pearson et al., in prep). However, during the pre-restoration survey, the floodplain in the reach and the majority of the lower catchment had been converted to permanent pasture. Watermeadow management practices have been observed to attenuate fine sediment loads by trapping material on the floodplain within long grasses and improve bank stability (Cutting et al., 2003; Micheli and Kirchner, 2002). It is conceivable that a decline in managed watermeadows may have affected the volume of fine sediment stored on the floodplain, and consequently, an increased volume of sediment may have been stored with the channel.

The pre-restoration morphology at the sub-reach scale exhibited a greater variability and morphological complexity than when assessed at the reach scale. Whilst the factors discussed above are likely to have reduced the diversity of form within the reach which contributed to the pre-restoration morphology, more local factors may explain some of the small variations revealed by the moving window analyses. The downstream pool was retained as a distinctive feature despite the Rother Navigation plans indicating that it may have been significantly widened in the late 18th Century. This is the widest part of the reach at ~30 m in width, yet this feature appears to have resisted significant depositional processes prior to the pre-restoration survey. However, possibly as a result of this widening, the cross-sectional area of the channel appears to be significantly reduced upstream of the downstream pool by a submerged bar (see Fig. 5.16 between cross-section D and E). The reduced cross-sectional channel area potentially explains the local increase of velocities in this area. Thus, the local modification of velocity patterns may have ensured fine sediments were flushed from the downstream pool at low flows. Potentially as discharge increased, the lateral convergence of flow upstream of the pool may have incited turbulence within the pool which may have further promoted scour processes and pool-maintenance (Clifford, 1993). Therefore, these observations could suggest that the downstream pool is an example of a forced pool (Thompson and McCarrick, 2010).

A localised increase in elevation was also observed approximately 40m along the channel from the start of the reach; according to the moving window analyses, the highest minimum bed elevation within the reach was observed in this area. In this location, a small embayment (most probably created through trampling by livestock) was observed with gentler bank slopes than the adjacent areas of the channel (Fig. 5.16 (see cross-section B) & Fig. 5.17). It is possible that a localised increase in width may have promoted deposition in this location, thus explaining the local increase in elevation in this area. This local variability in channel width may also offer an explanation as to why the bed in the area in which the riffle feature was later constructed was more variable than the rest of the channel. Therefore, these results highlight that sub-reach scale analyses (such as moving window analyses) can provide evidence to interpret local morphological processes which may need to be accounted for within river restoration design.

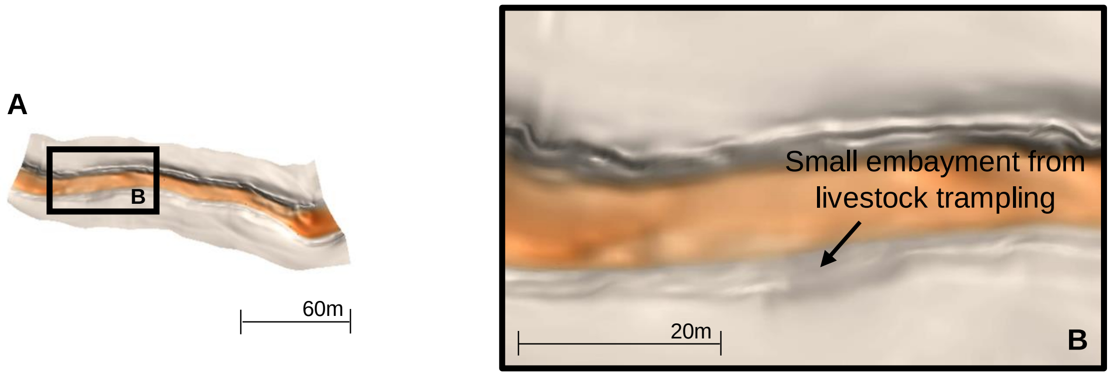


Figure 5.17 3D view of the DEM (A) highlighting a small embayment and localised increase in channel width (B) which may have locally influenced sediment transport processes.

5.4.2 Post-restoration geomorphology

Geomorphological processes prior to the 2013/14 flood events

The construction of the riffle feature instantly improved the morphological variability over the bed and the diversity of velocity patterns within the reach at low flows. This would suggest that the construction of the riffle feature was successful in improving the geomorphological performance of the reach. However, analyses at the feature scale actually suggest a loss of topographic complexity following restoration. Whilst the analyses of velocity patterns at this scale demonstrated some improvement, this observation highlights the importance of scale in the interpretation of the geomorphological performance of river restoration schemes.

The riffle feature increased the upstream water surface elevation by ~0.2 m, and generally increased the water surface slope over the riffle feature, however, this effect was most noticeable over the crest (Fig. 5.18). This would imply that the feature was functioning in a similar manner to a naturally formed riffle, as they have been observed to back-up flow like a broad crested weir (Clifford et al., 2002). It is likely that the reduced channel cross-sectional area (through the addition of material) combined with the increase in water slope resulted in the observed increased velocity over the riffle feature. Unlike a naturally formed riffle feature, the riffle crest would typically be found over the head of the feature and not the tail. Therefore, an increase in water surface elevation may be expected over the area upstream of a riffle but not over the riffle itself.

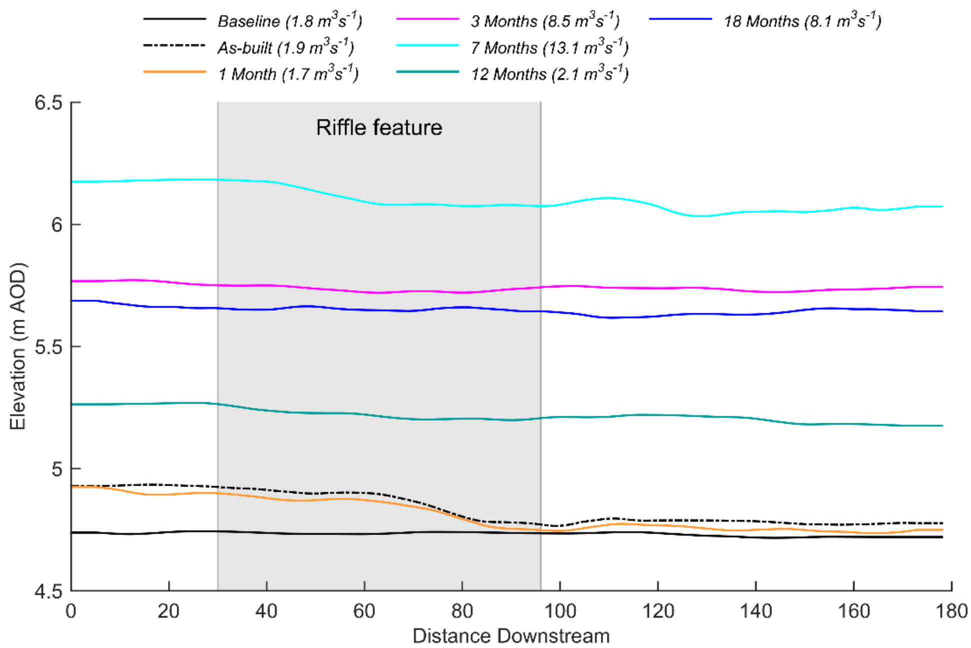


Figure 5.18 Water surface elevations throughout the reach from each survey.

Over the crest of the riffle feature, the velocities were observed to be at their maximum within the reach, and this area also exhibited a strong downwelling (Fig. 5.19). However, the tails of naturally formed riffles are often characterised by upwellings due to hyporheic exchange processes (Tonina and Buffington, 2009; Hassan et al., 2015; Mathers and Wood, 2016). Therefore, the riffle feature as designed was not fully mimicking the geomorphological processes of a naturally formed riffle. This interpretation of geomorphological processes was directly informed by the 3D velocity measurements captured by the ADCP. If traditional surveying techniques had been used during the monitoring survey, such as 2D current meters, this process may have been overlooked. Thus, this highlights the utility value of the ADCP for evaluating the geomorphological performance of river restoration schemes.

Immediately downstream of the riffle crest, an eddy was identified near the right bank following the lateral constriction of flow (Fig. 5.20). Similar flow patterns are characteristic of pools, whereby flow is constricted in pools to form a ‘jet flow’ and recirculating eddies are formed near the outer bank (Schmidt, 1990; MacWilliams et al., 2006; Thompson and McCarrick, 2010). Therefore, scour may have occurred in this location over time due to a potential localised increase in turbulence (Clifford, 1993). Further downstream of the riffle feature, no significant changes to the velocity patterns were immediately observed. As a result, this could suggest that the pool maintenance processes which may have been sustaining the downstream pool (as discussed in section 5.4.2) were unlikely to have been immediately modified during low flows by the construction of the riffle feature.

Minimal changes to the morphology and velocity patterns suggest that the riffle feature was resilient to both significant depositional and erosional processes during low flows

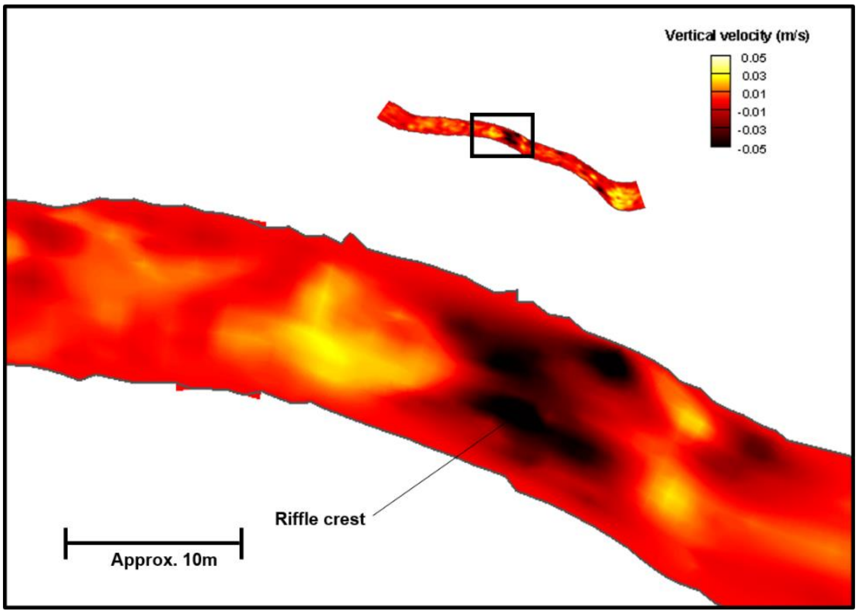


Figure 5.19 A vertical velocity plot showing downwelling over the riffle crest

within the first month following construction. The results suggested that within the first 3 months following construction, the overall form of the riffle feature was stable up to bankfull flows. The 3-month post-construction survey was captured during moderate flow conditions and in contrast to the low flow surveys, the effect of the feature on the water surface slope was negligible (Fig. 5.18). Thus, this could suggest the feature had drowned out, a typical characteristic of natural riffles. The drowning out effect has also been observed during laboratory evaluations of artificial riffle designs that have been implemented on the North Fork of the Chicago River, US (Rodriquez et al., 2013). Interestingly, there appear to be very few documented evaluations of artificial riffles that have been installed as part of restoration schemes. This is a significant observation on the state of learning in riffle design and construction, and more generally within river restoration.

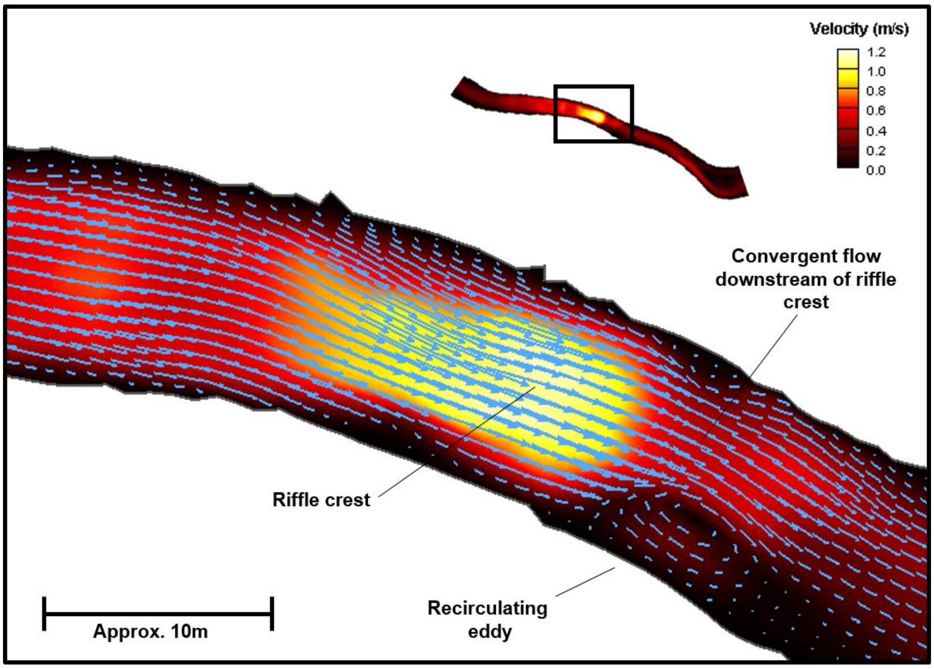


Figure 5.20 Vector plot of the immediate post-construction survey over the riffle crest.

A degree of convergence in velocity with discharge was observed along the reach, potentially due to a smoothing of the water surface slope and a greater hydraulic efficiency as the riffle became drowned out. In unmodified channels with pool-riffle sequences, a convergence or (very rarely) a reversal of velocity has been observed between the two distinct forms (Keller et al., 1971; 1972; Clifford and Richards, 1992; Clifford, 1993; Thompson et al., 1999; Strom et al., 2016). However, the velocities over the riffle feature were still significantly higher than in other areas of the reach during both moderate and low flows. It is possible that the persistence of higher velocities over the riffle feature is due to a variation in cross-sectional area along the reach (Caamaño et al., 2009; de Almeida and Rodriguez, 2011). The cross-sectional area was reduced through the construction of the riffle feature and the channel was also much wider upstream and downstream of the riffle feature. Therefore, the velocity may have increased over the riffle feature to ensure the continuity of flow along the reach. Whilst the riffle feature may not have been fully mimicking the processes of a pool-riffle sequence at a moderate flow, it was successful in promoting hydraulic variability within the reach. Still, if the performance of the scheme was solely evaluated at the feature scale, the effectiveness of this feature in promoting hydraulic diversity could be undervalued as the distribution of the velocities became more uniform over the riffle feature.

Despite elevated velocities over the riffle feature, a small amount of deposition (up to 0.1 m) over the riffle feature suggests that it may not have been fully resilient to depositional processes during flows up to bankfull. Given the impediments to coarse sediment transport within the catchment (Chapter 3), it is highly likely that this material deposited over the riffle feature largely comprised of sand. It is conceivable that this deposition may have been promoted by the riffle feature itself as the amount of sediment transported through the reach seasonally increased (autumn-winter). As riffles can influence the water depth and slope upstream of the crest, the design of the riffle feature with the crest sited at the downstream end could have reduced the capacity to transport material over the length of the feature during lower flows. This could also explain the scour observed downstream of the feature in a 'hungry water' effect, as cleaner and potentially more turbulent flows could have an increased capacity to transport material downstream of the feature (Kondolf, 1997). These observations could imply that the riffle feature (as designed) was not completely self-cleansing and resilent to seasonal increases in sediment load.

The velocity patterns over the downstream pool exhibited a more complex pattern at moderate flows than observed during low flows (Fig. 5.21), as suspected from processes interpreted from the low flow surveys. The velocity vectors suggest that flow is concentrated through the centre of the pool with a recirculating eddy observed near the outer bank. This may suggest that the effect of the constriction on directing the core of

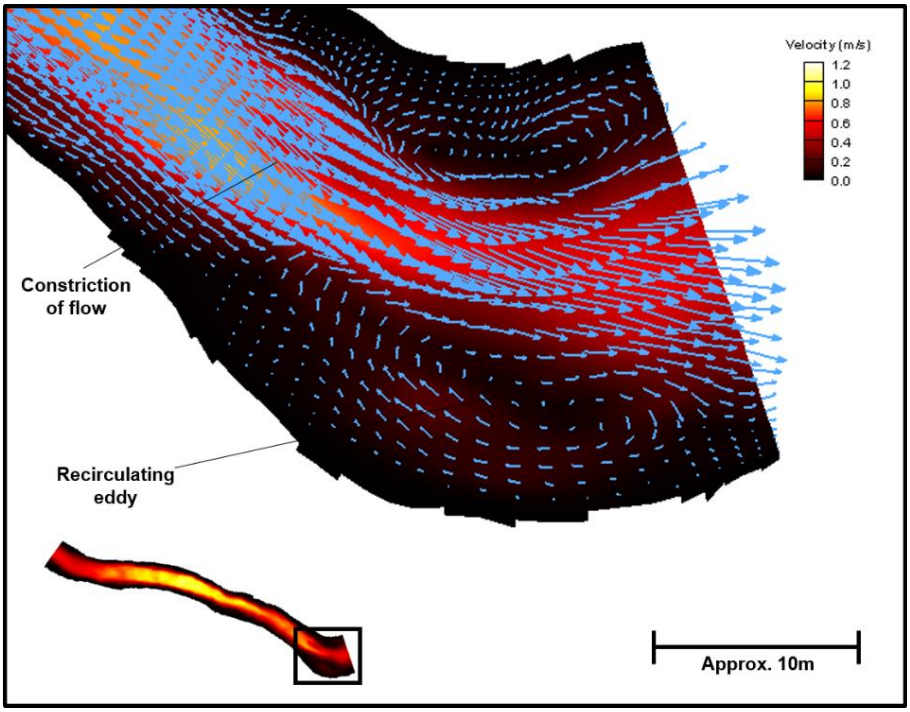


Figure 5.21 Depth average velocity vectors for the 3-month post-construction survey over the downstream pool

maximum velocity was less prominent at higher flows. The observation of the eddy tied in with an area of localised scour which may suggest that near-bed turbulence during higher flows contributed to the maintenance of the downstream pool (Clifford and Richards, 1992; Clifford, 1993). However, the processes within the lower reach were unlikely to have been heavily influenced by construction feature. These observations serve to highlight the importance of accounting for and anticipating form-scale processes as well as catchment scale geomorphological processes in river restoration design.

Geomorphological processes following the 2013/14 flood events

Within the 3-7 month period following the construction of the riffle feature, multiple out of bank flows were experienced within the reach as a result of the 2013/14 flood events which left the study site inaccessible for most of January and February 2014. The results in Section 5.2 indicated that significant morphological changes were detected within the reach compared with both the immediate and 3-month post-construction surveys. One of the significant morphological changes observed immediately following the 2013/14 flood events was the re-location of the riffle crest from the tail to the head of the riffle feature. This structural modification to the riffle feature is not completely unsurprising as the upstream location of the topographic high imitates the form of riffles in natural rivers. In the 3-7 month post-construction period a pool formed immediately upstream of the riffle feature, but there was no clear pattern of divergence during the near bankfull 7-month post-construction survey (Fig. 5.22). This contrasts within MacWilliams et al. (2006) who suggested the lateral divergence of velocity upon the exit of pools (which promotes the deposition downstream of the pool feature) contributes to the maintenance of riffle features.

Some gravel bed rivers are intentionally widened locally to utilise this natural process to restore riffle features (Weber et al., 2009; Brown et al., 2016). Therefore, it is not completely unsurprising that the area of maximum deposition over the riffle feature coincided with the area of the channel that had been locally widened by livestock induced bank erosion. The baseline survey suggests that depositional processes were conceivably working in this area prior to restoration (as discussed in Section 5.4.1). Therefore, this process and potential impact on the structural integrity of the feature may have been identified if the baseline dataset had been available during the design phase. Consequently, this serves to highlight the importance of undertaking a full high-resolution geomorphological baseline survey within the initial stages of the restoration project to ensure that the restoration design complements natural processes.

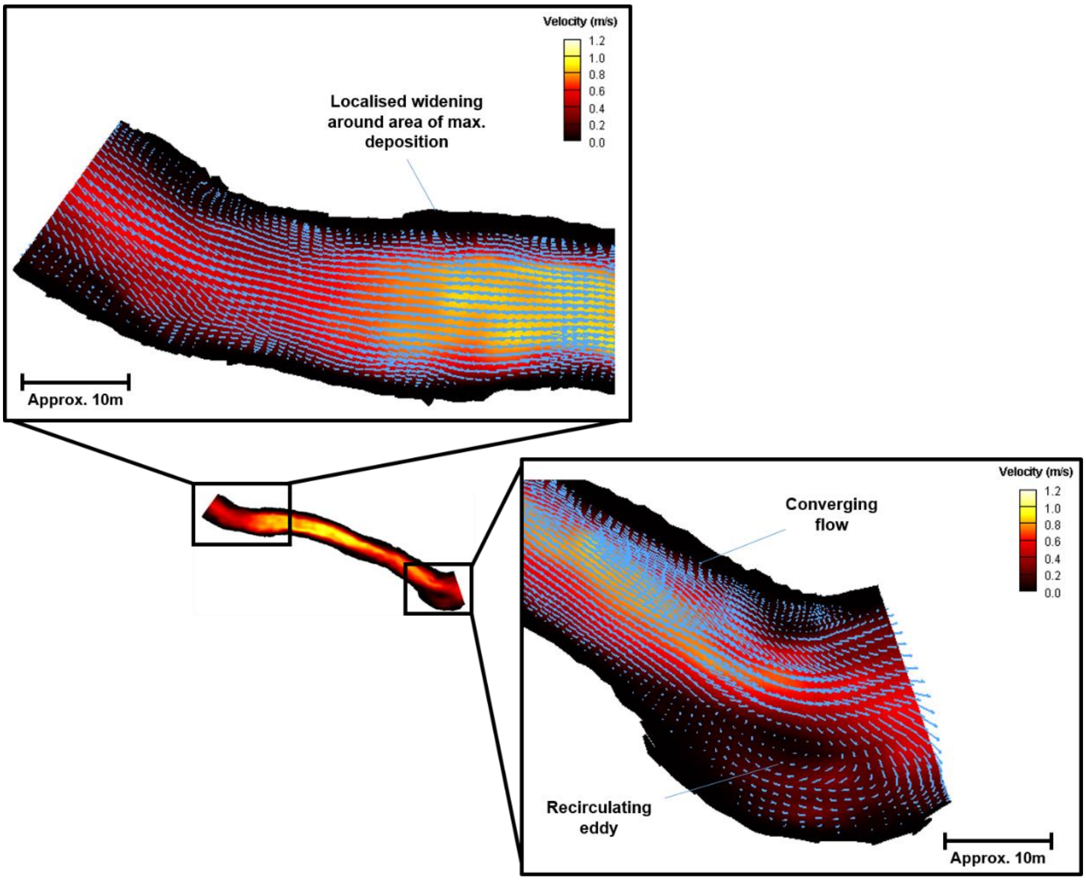


Figure 5.22 Depth average velocity vectors for the 7-month post-construction survey highlighting the upstream and downstream pool

Following the flood events, the riffle feature gained a significant amount of material until 12-months post-construction. Spot samples from the surface of the head of the riffle feature (the site of maximum fill) taken during the 12-months post-construction survey were largely composed of mixed gravel (Fig. 5.23). Some fine sediment was also found within the sample which could indicate a degree of sedimentation over the riffle feature, but it is important to note that the sample was probably taken from the cleanest part of the riffle as material typically fines downstream over riffles (Mathers and Wood, 2016). It is likely that samples from the tail of the riffle feature would exhibit a larger proportion of fines, but significant deposition over the tail of the feature was not observed upon inspection during low flows. The rate of fine sediment infiltration into the gravels of the riffle feature would have likely increased during (and following) the 2013/14 flood events due to an increase in sediment supply (Sear, 1993). The widespread observation of gravels over the rifle feature following the flood events was encouraging from a resilience point of view. Nonetheless, a separate study that monitored the riffle feature indicates that there was a significant potential for the storage of fine sediment within these gravels (Evans et al., 2017).

The presence of gravel on the surface of the feature might lead to the suggestion that a supply of gravel to the riffle feature was locally available. However, given the significant barriers to sediment transport and local geology as discussed in Chapter 3, this seems unlikely. Therefore, an alternative hypothesis could be that the gravel feature has been a sediment trap. Fine sediment has been observed migrating laterally and horizontally through gravel at the subsurface and consequently accumulating in the interstitial spaces of the gravel (Carling, 1984; Petticrew et al., 2007; Koiter et al., 2015; Mathers and Wood, 2016). Further sedimentological investigation would be required to fully test this hypothesis, but it would go some way to explaining how gravel is still observed at the surface of the feature.

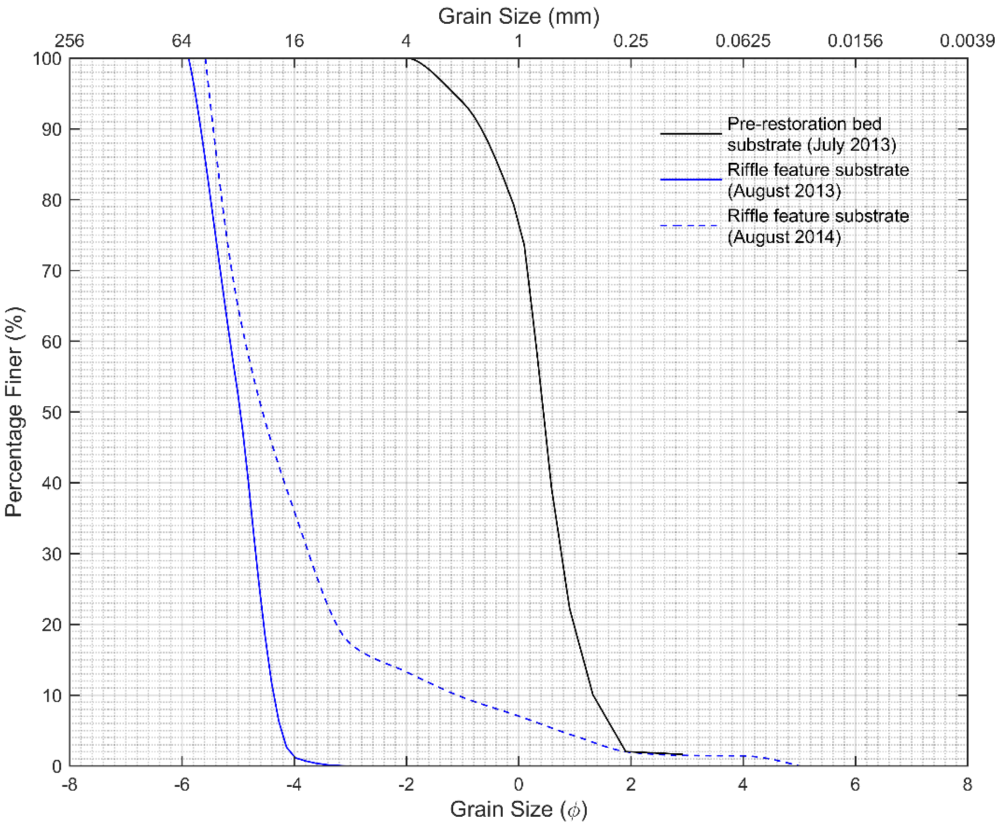


Figure 5.23 Particle size distribution of sediment samples from the reach before, after and 1 year following construction of the riffle feature.

High flow events have been observed to flush sediment from the subsurface interstitial spaces of gravel (Petticrew et al., 2007). This process of infiltration of sediment may also explain some of the seasonal blanket scour and fill observed over the feature following the 2013/14 flood events. In the assessment of the effectiveness of artificial sediment traps, traps comprising washed gravel were found to be almost twice as effective at trapping sediments as those comprising unwashed gravel. This was due to the former having larger interstitial spaces through artificial packing (Petticrew et al., 2007). Therefore, in the period following construction, the riffle feature would have been a prime sediment trap as the mudstone cobbles and gravel were not sorted on placement during construction. If this hypothesis were explored further, the use of such features as sediment traps could be used as a fine sediment mitigation option. However, the habitat value of such features may be limited by the fine sediment accumulation.

The significant scour upstream of the feature defined a deep pool and the downstream pool also deepened. Similar process were observed over the downstream pool during the near bankful survey (7-month survey) to those observed before the 2013/14 flood events. This was a migration of core velocity from outer to the inner-bank as the effect of the constriction was likely to have been drowned out, and the formation of an eddy towards the inner bank (Fig. 5.22). It is possible that this feature was maintained and deepened during the flood events through turbulence processes (Clifford and Richards, 1992; Clifford, 1993). During the 12-month post-construction low flow survey, the velocity patterns exhibited similar characteristics to the pre-flood and baseline datasets (Fig. 5.24). This might suggest that the processes operating within the downstream pool were unaffected by the construction of the riffle feature. However, this is an interesting contribution to the discussion on the scale at which river restoration monitoring should be performed. If these processes were occurring independently of the riffle feature but increased the morphological diversity over the wider reach, the inclusion of this area within the results may over-exaggerate the influence of the riffle feature. This example highlights the importance of the following in identifying river restoration performance:

- the quantification of 3D velocity patterns at a high resolution and at multiple flows;
- undertaking a before and after (BA) assessment and for identifying pre-existing morphological processes to fully, and;
- the analysis of restoration performance at multiple spatial scales (dependent on the scale of intervention) to identify the influence of direct channel modifications.

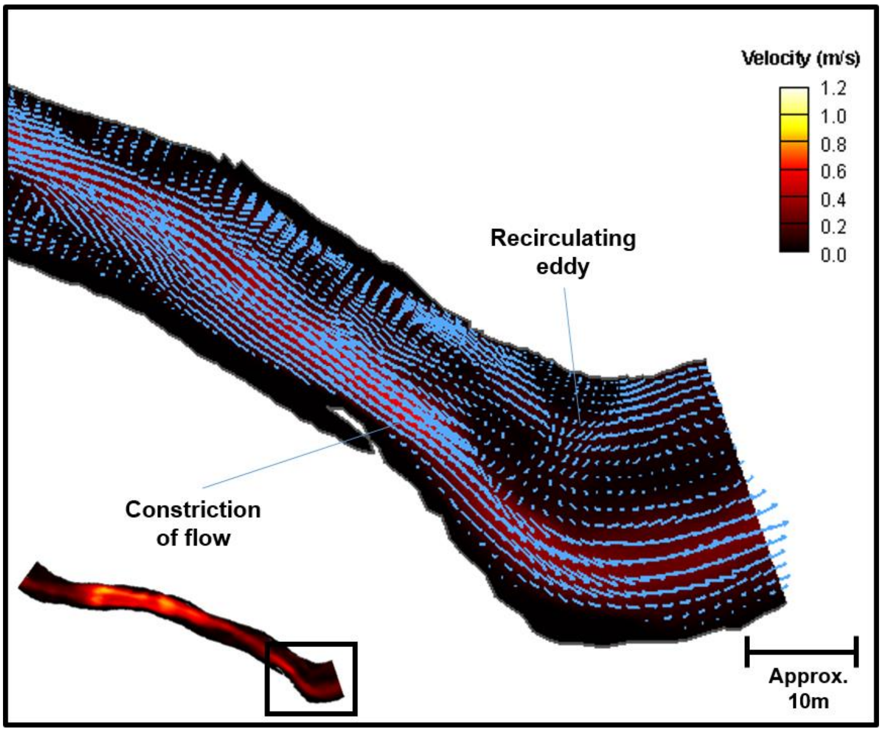


Figure 5.24 Vector plot of the 12-month post-construction survey over the downstream pool

The creation of the upstream pool does not appear to be associated with any significant local lateral convergence of velocity patterns (Fig. 5.22). The post-flood velocity patterns suggested only a slight convergence in velocity patterns along the reach, with the structurally modified riffle feature maintaining its influence on velocity patterns at higher flows. However, the water surface profile during the near bankfull survey (7-months post-construction) exhibited more variation than the lower flow surveys post-flood. A possible explanation for this is an increase in water depth upstream of the riffle feature (Fig. 5.18), which may have been caused by an interaction between the flow and complex bank morphology during higher discharges. The localised increase in water depth (possibly due to the backing-up of flow) may have increased the shear stress and the capacity to entrain sediment, thus leading to the creation of the upstream pool (Milan et al., 2001; Jackson et al., 2015). Interestingly, there did not appear to be a distinct difference in the amount of deposition between the two pools within the reach as the flood flows receded, despite there being no immediately obvious processes operating at low flows to ensure the maintenance of the recently created upstream pool.

The structure of the riffle feature was intentionally designed to be stable during flows up to bankfull. However, the structural modification detected through the monitoring programme suggests that the design of the feature was reworked during flows in excess of bankfull. Nonetheless, the general maintenance of riffle feature suggests it is somewhat resilient to extreme events. This observation may also suggest that given the appropriate substrate, the river may recover a more natural longstream form over time and thus time (and money) spent during the construction of riffle features could be reduced. However, significant formative flows may take several years to occur, unless an appropriate range of flows may be augmented (e.g. by reservoir releases) without risking adverse flooding effects. In this situation, a form with the crest sited over the riffle head is likely to be more sustainable and more likely to self-cleanse. The post-flood riffle feature with the reworked upstream crest appeared to have a less significant impact on the water surface slope within the reach at low flows than the feature did immediately post-construction (Fig. 5.18). This could explain the lowering of the peak velocities within the reach following the flood events which were once sited over the crest of the riffle feature. Nonetheless, after the flood events, the riffle feature was still providing an improved geomorphological performance within the reach.

5.5 Summary of key findings from the geomorphological performance evaluation of the RHES

In summary, prior to the 2013/14 floods, the construction of the riffle feature improved the morphological and hydraulic variability within the full study reach, without compromising existing morphological processes. The form of the feature was reasonably resilient in flows up to bankfull, but the results suggest it may not have been fully self-cleansing as a potential result of the siting of the crest. The riffle feature was also reasonably resilient to extreme events and following structural modification of the feature during the 2013/14 floods it appeared to be functioning (and shaped) more akin to a natural riffle. It is not clear from the results if improvements in morphological diversity within the reach were solely down to the riffle placement or in combination with the 2013/14 flood events. Thus, this highlights that a control site (Chapter 2) may have been beneficial in resolving this uncertainty had the lead in time prior to construction been longer for this study (Chapter 4).

Whilst locally the feature may have improved the geomorphological performance within the study reach, there is limited evidence to suggest that the feature had a significant impact beyond the reach. This was not a failing of the riffle feature as it was designed as a reach scale restoration effort, however, as discussed in Section 5.4.1, many of the processes affecting the geomorphological performance were operating at the catchment scale. Therefore, the ability of the riffle feature to be robust to catchment-scale processes over a longer time-period without further intervention is uncertain. As a result, recommendations for future management on the River Rother (and in similar catchments), based on the performance of the riffle feature throughout the 18-month monitoring period would be:

- to explore catchment scale approaches to manage the in-stream fine sediment load, which may include sediment traps, riparian buffers and the restoration of floodplain connectivity;
- to restore longitudinal sediment connectivity and consider frequent gravel augmentation to sustain riffle features within the lower catchment, if desired;
- to consider trialling a riffle feature with the crest sited over the head of the riffle feature, as this is more likely to be self-cleansing, or (if feasible) a range of designs with the view of optimising restoration performance, and;
- to consider the installation of a deflector or similar constriction mechanism just upstream of the reach to try to maintain the newly created upstream pool.

This chapter has revealed some key findings which are of significance not only for the future management of the River Rother but for the practice of wider river restoration monitoring design and river restoration monitoring. The salient findings for monitoring practice can be summarised as follows:

- Undertaking baseline surveys and wider consideration of geomorphological processes are critical for identifying suitable river restoration objectives and informing sustainable in-channel river restoration designs that work with natural processes;
- The ADCP data can be used effectively to identify both local and catchment scale geomorphological processes that may be influencing a specific reach (e.g. seasonal sediment transport processes);
- The importance of considering an appropriate spatial scale for the collection of monitoring datasets and their assessment of geomorphological performance;
- Without the use of control sites, it can be difficult to distinguish between change resulting from intervention and natural processes; and,
- A velocity convergence but not reversal was observed between this constructed riffle feature and the adjacent pools as discharge increased up to bankfull. Significant out-of-bank flows were experienced within the reach during the monitoring period (but not captured due to health and safety risks) during which significant geomorphological change occurred. Consequently, it would be

beneficial to utilise remotely operated technologies to capture data at these flows to develop a better understanding of pool-riffle maintenance mechanisms.

The next chapter will report on the physical habitat performance of the RHES and make broad recommendations for the future management of the River Rother which are both geomorphologically and ecologically aligned.

6 River Rother Habitat Enhancement Scheme: Physical Habitat

6.1 Introduction

The overall aim of this chapter is to demonstrate the applicability of ADCP data for assessing physical habitat performance of river restoration schemes. To achieve this, the chapter uses the geomorphological monitoring data from the River Rother Habitat Enhancement Scheme (RHES) to provide an assessment of physical habitat performance over the 18-month monitoring period. This chapter reports on a range of techniques which were adapted from the literature reviewed in Chapter 2. The RHES performance assessment will be used to add to the geomorphological recommendations (Chapter 5) for future management strategies of the River Rother and similar environments. The contents of this chapter also provide a platform for a discussion in Chapter 7 on the importance of considering the resolution and spatial dynamics of physical habitats within river restoration monitoring schemes.

6.2 Results: Physical Habitat heterogeneity

Two assessments of habitat heterogeneity are presented in this section, namely; a hydromorphological index of diversity (HMID) assessment and hierarchical cluster analysis (HCA) (Chapter 4). The HMID scores suggest that physical habitat heterogeneity (PHH) at low flows increased over the full study reach following restoration as observed in the as-built, 1-month and 12-month post-construction surveys (Fig. 6.1 and Table 6.1). Following the 2013/14 flood events, the HMID scores suggest that PHH at low flows improved further still on the immediate post-restoration condition. Interestingly, when considering only the riffle feature this trend was reversed. The HMID scores decreased slightly immediately following restoration and further following the 2013/14 flood events at low flows. However, the magnitude of these changes were not as significant as the changes observed over the wider reach. When comparing the pre-restoration HMID scores over the full study reach and the area in which the riffle feature was later constructed (sub-reach), the results suggest that prior to restoration physical habitat over the sub-reach was more homogenous than the wider reach. Nonetheless, the trend of decreasing HMID scores may indicate that PHH of the sub-reach deteriorated at low flows following restoration.

The downstream moving window analyses (MWA), however, gives a slightly contrasting interpretation of PHH at low flows when considering the more local scale variability of physical habitat. The HMID scores for the pre-restoration condition of the channel were relatively constant throughout the reach when compared to later surveys. However, the area in which the riffle tail was later constructed and the downstream pool had a higher HMID score than elsewhere within the reach. Immediately following restoration during low flows, minimal changes to the HMID score were observed in the areas that were not physically altered by restoration activities. This is not completely surprising as minimal changes were observed to the morphology and velocity patterns in these areas immediately following construction of the riffle feature. Conversely, the HMID score increased over the constructed tail of the riffle feature but decreased over the head of the riffle feature. This could indicate that the micro-habitat heterogeneity improved over the riffle crest (tail) following restoration, however, some of the PHH over the riffle head was lost. This is an interesting observation, as when assessing the entire riffle feature, the results suggest the area had become more homogenous following restoration.

Table 6.1 HMID scores

Survey	Reach HMID	Riffle feature HMID
Baseline	6.0	5.1
As-built	8.6	4.9
1 Month	8.9	4.7
3 Months	6.3	3.9
7 Months	6.1	3.6
12 Months	10.8	4.5
18 Months	7.3	3.6

Following the 2013/14 flood events, the MWA of HMID suggests that PHH during low flows improved significantly both upstream and downstream of the riffle feature. Over the tail of the riffle feature (previously the riffle crest), the HMID values suggest that PHH decreased as a potential result of peak velocities being lost as the sediments reworked during the floods. In contrast, the HMID scores increased over the head of the riffle feature (the newly sited crest) suggesting an improvement in PHH. However, the newly sited riffle crest was not as heterogeneous during low flows as the riffle crest immediately following construction.

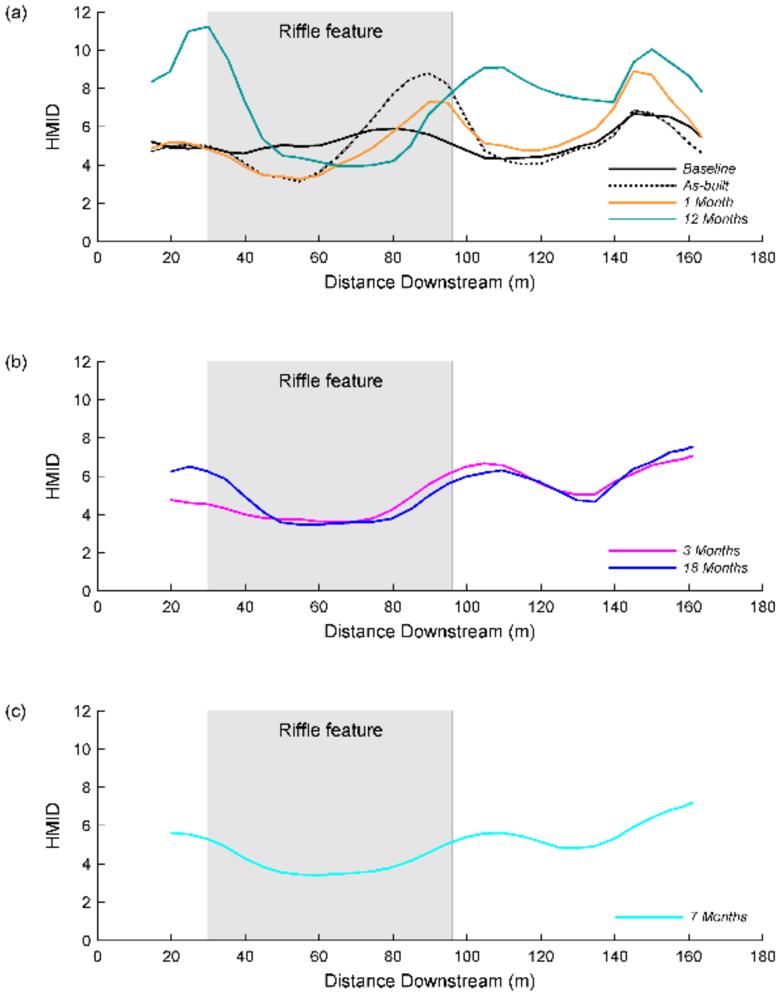


Figure 6.1 Moving window analysis of HMID, (a) low flow surveys, (b) moderate flow surveys and (c) high flow survey.

Neither a moderate or high flow were captured prior to restoration, therefore, it is not possible to compare the 3, 7 and 18-months post-construction surveys to a baseline condition. However, the two moderate flow surveys (3 and 18 months) fall before and after the 2013/14 flood events. Thus, these can be used compare the pre- and post-flood physical habitat performance of the scheme up to a moderate flow. The 7-month post-construction survey was undertaken during a high flow following the 2013/14 flood events. Therefore, the scheme's physical habitat performance following the flood events can be discussed up to a high flow. Prior to the 2013/14 floods, the HMID scores were significantly lower during a moderate flow over the entire reach and riffle feature than observed during low flows (Table 6.1). The HMID scores of the MWA suggest a loss of PHH upstream and over the riffle feature, yet an increase in PHH downstream of the riffle feature. This is potentially a result of the morphological diversity gained (through scour) in this area during the first 3 months following construction.

Following the 2013/14 flood events, the HMID scores were higher over the full study reach but slightly lower over the riffle feature when compared to the pre-flood moderate flow. In comparison to the post-flood low flow, the HMID scores were significantly lower over the full study reach and the riffle feature during the moderate flow and high flows. However, there was a less marked difference between observations of the PHH between the moderate and high flows. The MWA largely echo the trends observed thus far, however, a notable observation is a significant increase in HMID over the upstream pool during moderate flows following the 2013/14 flood events. Again, this is a possible result of the morphological diversity gained in this area of the channel.

The results of the hierarchical clustering analysis (Fig. 6.2, Table 6.2) also indicated low PHH prior to restoration. Three distinct hydraulic patches (B1, B2, B3) were detected within the reach, of which Patch B1 occupied ~14% of the channel area and was fragmented. Patch B2, which occupied ~37% of the channel (largely in the margins), also exhibited fragmentation. However, Patch B3, the dominant patch accounting for ~49% of the channel, was more coherent and occupied most of the central areas of the channel. Immediately following, and 1 month post-construction, the diversity in hydraulic patches increased with the detection of 4 distinct hydraulic patches. One of these patches (Patch AB2 & 1M2) occupied a very small area (< 5%) of the channel over the constructed riffle crest (riffle tail). However, overall the hydraulic patches were more similar in size during low flows than prior to restoration.

Patches AB1 and 1M1 which occupied the channel margins, were more fragmented than the previous patch detected in this area of the channel. Patches 3 (AB3 & 1M3) and 4 (AB4 & 1M4) occupied areas in the centre of the channel, with the latter occupying areas of velocity that were increased through the installation of the riffle feature and exhibiting as a coherent space. Patches AB3 and 1M3 which largely occupied areas that remained unchanged compared to their pre-restoration morphology and velocity, appeared to be more fragmented than patch B3. However, AB3 and 1M3 were comprised of many smaller fragmented areas and some larger but more coherent spaces, which offers an explanation for the lower edge ratio observed for these patches. Given the transitional patches largely retained their coherence following restoration during low flows, the loss of ~5-10% transitional habitat was potentially beneficial in improving PHH.

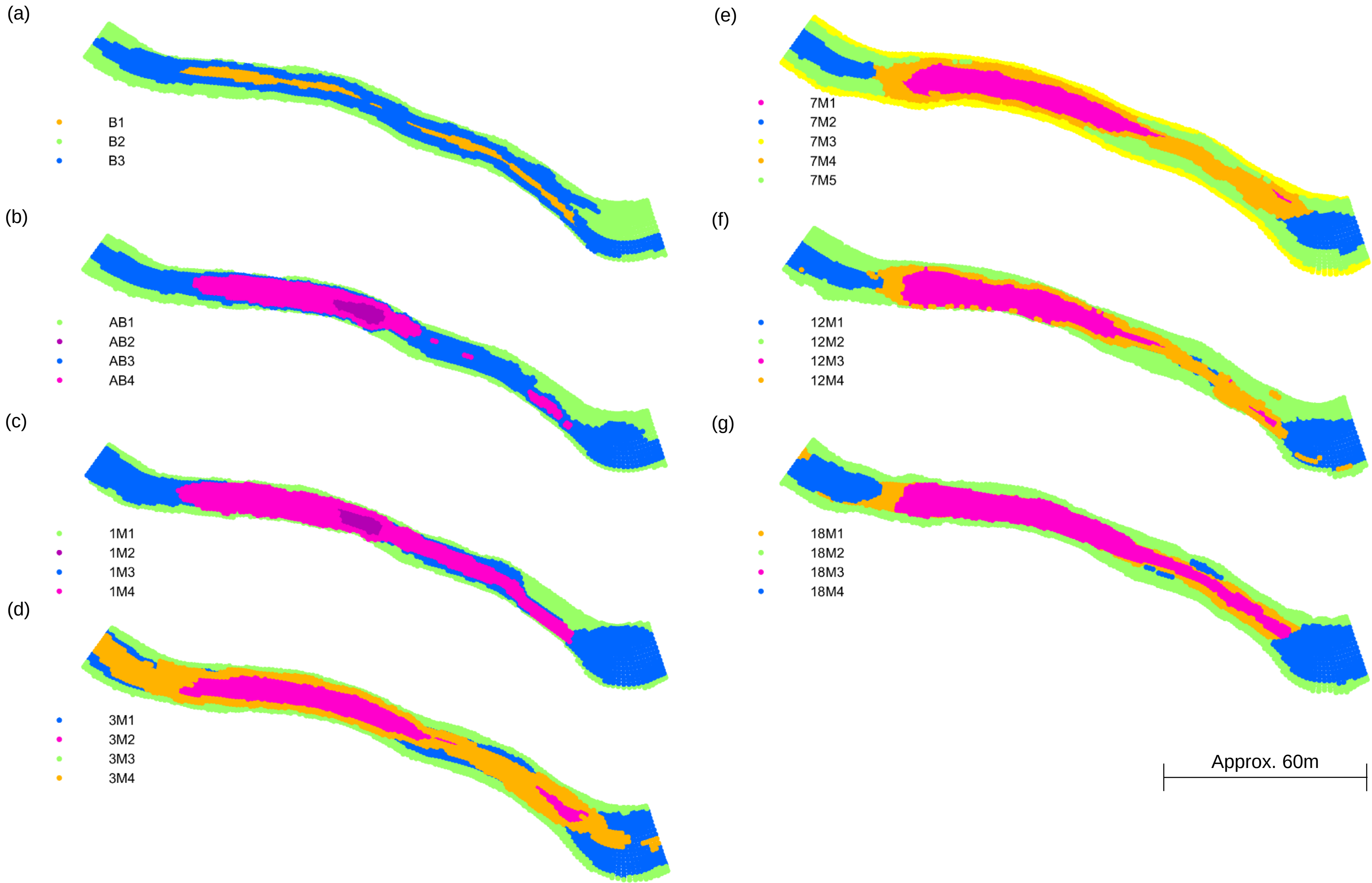


Figure 6.2 Hierarchical cluster map of distinct hydraulic patches (a) pre-restoration survey, (b) as-built survey, (c) 1-month post-construction survey, (d) 3-months post-construction survey, (e) 7-months post-construction survey, (f) 12-months post-construction survey and (g) 18-months post-restoration survey.

Table 6.2 Results of hydraulic clustering analysis

	Patch Metrics				
Patch Code	**Character**	**% Area**	**Divisions**	**Edge Ratio**	**Fractal Dimension**
Baseline (low flow)	***Simpsons Index of Diversity = 0.60***				
B1	Transitional	14.26	2	1.01	1.50
B2	Margins	36.70	7	0.98	1.56
B3	Transitional	49.04	1	0.55	1.41
As-built (low flow)	***Simpsons Index of Diversity = 0.66***				
AB1	Margins	31.16	9	1.25	1.62
AB2	Riffle crest	4.12	1	0.58	1.07
AB3	Transitional	44.18	9	0.57	1.40
AB4	Riffle	20.53	2	0.65	1.37
1 Month (low flow)	***Simpsons Index of Diversity = 0.67***				
1M1	Margins	20.73	16	1.79	1.72
1M2	Riffle crest	3.38	1	0.61	1.05
1M3	Transitional	40.42	6	0.46	1.33
1M4	Riffle	35.46	1	0.52	1.35
3 Months (mod. flow)	***Simpsons Index of Diversity = 0.73***				
3M1	Pool	23.81	6	0.76	1.46
3M2	Riffle	25.93	2	0.42	1.27
3M3	Margins	16.25	8	2.20	1.79
3M4	Transitional	34.01	5	0.80	1.50
7 Months (high flow)	***Simpsons Index of Diversity = 0.80***				
7M1	Riffle	23.46	2	0.36	1.23
7M2	Pool	15.35	2	0.36	1.18
7M3	Margins	18.88	3	1.72	1.72
7M4	Transitional	23.38	3	0.82	1.49
7M5	Margins	18.93	7	1.01	1.54
12 Months (low flow)	***Simpsons Index of Diversity = 0.73***				
12M1	Pool	25.15	4	0.38	1.24
12M2	Margins	33.61	7	0.89	1.54
12M3	Riffle	26.96	2	0.36	1.24
12M4	Transitional	14.28	9	0.89	1.47
18 Months (mod. flow)	***Simpsons Index of Diversity = 0.75***				
18M1	Transitional	21.57	12	0.88	1.50
18M2	Margins	25.37	6	1.41	1.67
18M3	Riffle	28.78	1	0.44	1.31
18M4	Pools	24.28	3	0.30	1.17

Following the 2013/14 flood events, 4 distinct hydraulic patches were again observed during the 12-month post-construction survey. There was an increase in equality between patch areas in comparison to patch areas observed prior to and immediately following restoration. The dominant patch at this time (Patch 12M2) was observed in the channel margins. This patch exhibited fewer divisions, a lower edge ratio and reduced shape complexity than patches observed previously in the channel margins. This may indicate that the marginal physical habitat in the reach increased in both coherence and abundance on both the pre-restoration and immediate post-construction condition. The hydraulic patches observed over the pools within the reach (Patch 12M1) and over the core of the riffle feature (Patch 12M3) exhibited as highly coherent and distinctive patches. Patch 12M4, however, was more fragmented but associated with transitional areas between patches relating to typical morphological forms (i.e. pools, margins and riffle feature). This was the least abundant and most fragmented patch observed. Thus, these observations indicate that PHH improved following restoration, and that hydraulic patches have become more distinct at low flows following the 2013/14 floods.

The diversity of hydraulic patches at moderate flows was improved as discharge increased, both pre- and post-flood. Four distinct hydraulic patches observed during the 3 and 18-month post-construction survey, which was the same as that observed immediately following construction of the riffle feature. However, at moderate flows the area of the channel occupied by each patch was more equal, particularly following the flood events. Furthermore, all hydraulic patches except the transitional patch observed during the 18-month post-construction survey were less fragmented than those detected during the 3-month post-construction survey. For example, the marginal hydraulic patches in both surveys (3M3 & 18M2) were highly fragmented, but significantly fewer divisions of the patch were observed following the 2013/14 floods. These observations suggest that the PHH observed within the reach at moderate flows as constructed was not only maintained, but enhanced following the morphological changes occurring as a result of the 2013/14 floods.

The greatest diversity of hydraulic patches through the monitoring campaign was observed during the high flow survey (7-months post-construction). An additional hydraulic patch was detected during this survey, and all patches were relatively equal with respect to channel occupancy. The hydraulic patches detected over the riffle feature and the pools within the reach (7M1 & 7M2), exhibited a higher level of coherence. However, the hydraulic patches detected in the channel margins (7M3 & 7M5) were the most fragmented patches observed during this survey, yet were less fragmented than other patches detected in the margins during all other surveys. The HCA results may suggest that the high flow survey exhibited a greater potential for hydraulic complexity than low and moderate flows. However, there were no comparable flows to validate this observation for the pre-flood morphology of the reach.

6.3 Results: Physical Habitat Simulation

Habitat suitability, based on preference criteria, was simulated using velocity and depth measurements collected using the ADCP during the 18-month monitoring period of the RHES. These results also exhibited significant variability over the 18-month monitoring period for the simulated preferences of the chosen fish species; brown trout, dace and roach (as outlined in Chapter 4). This section is structured such that the results of each species are reported within a separate sub-section. Within these sub-sections the physical habitat performance for the life stages are reported with respect to their performance at either; prior to restoration, post-restoration at low flows or post-restoration at moderate and high flows. As data was collected during a low flow condition prior to restoration (as discussed in Chapters 4 and 5), it was only possible to compare the physical habitat performance of these species following restoration at low flows. Following restoration, comparable low and moderate flows were captured before and after the 2013/14 flood events which affords an opportunity to assess physical habitat resilience to extreme events.

6.3.1 Brown trout

Pre-restoration condition (low flow)

Brown trout was a target species of the RHES, the simulations suggest that prior to restoration physical habitat for most life stages of this species was limited at low flows. The simulations suggest that physical habitat for adult and spawning brown trout (which prefer a broad range of velocities of between 0.4 – 0.8 m s^{-1} and water depths of 0.25 – 1.5 m (Armitage and Ladle, 1991)) was highly suitable along the centre of the reach (Figs. 6.3 & 6.4). However, these physical habitats (particularly for spawning brown trout as they prefer slightly shallower environments than the adults) were suggested to be in a low abundance and moderately fragmented (Table 6.3). Similarly, the results suggest that physical habitats for fry, which prefer shallow water depths (less than 0.3 m) and slower velocities (less than 0.3 m s^{-1}) (Armitage and Ladle, 1991), were most common within the channel margins. The physical habitat for this life stage was simulated as the least abundant and most fragmented prior to restoration (Fig. 6.5). Furthermore, 74% of the simulated fry habitat was classified as of a low suitability (less than a 0.4 score on the Habitat Suitability Index (HSI)).

In contrast, the physical habitat for juvenile brown trout was predicted in high abundance along the reach prior to restoration. The preferred physical habitat for juvenile brown trout is a slightly shallower (0.25 – 0.8 m) and slower flowing (0.2 – 0.6 m s^{-1}) environment than that preferred by adult brown trout (Armitage and Ladle, 1991). The simulations suggest that 82% of the total suitable physical habitat for juvenile brown trout was highly suitable (more than 0.7 HSI score) and was highly coherent (Fig. 6.6 & Table 6.3). Therefore, these results suggest that the reach presented a high quality physical habitat for juvenile brown trout, but poor quality physical habitats for the remaining life stages during low flow conditions prior to restoration.

Post-restoration (low flows)

The simulation of physical habitat suitability for the immediate and 1-month post-construction surveys were very similar, as expected given the results of the velocity patterns and morphological change. The results suggest that physical habitat suitable for adult brown trout increased over the riffle feature (particularly across the channel) potentially due to an increase in velocities in this area. However, the results also suggested that physical habitat suitability decreased over the constructed crest (tail) of the riffle feature (Fig. 6.3). It is likely this decrease in suitability primarily resulted from a reduction in water depth in this area following the construction of the riffle feature. The velocities within this area (Chapter 5), whilst less suitable, were within the tolerable range for brown trout (Armitage and Ladle, 1991). The spatial metrics indicate that the highly suitable habitat was more abundant within the reach immediately following restoration (Table 6.4). Furthermore, an increase in the number of sub-patches but a decrease in edge ratio and shape complexity suggest that adult brown trout habitat was more coherent immediately following the construction of the riffle feature.

The simulations indicated that physical habitat suitable for adult brown trout increased after 12-months post-construction survey following the 2013/14 floods. This was particularly notable over the riffle feature, suggesting a further improvement on the pre-restoration physical habitat for adult brown trout. It is likely that this increase in physical habitat occurred as a result of the re-working of the riffle feature during the 2013/14 floods, which overall elevated velocities over the riffle feature despite the loss of the peak velocities (Chapter 5). However, the results indicate that this physical habitat also decreased in suitability in the area of extensive deposition over the head of the riffle feature (that formed the post-flood crest) as the depth of water decreased (Fig. 6.3). The results suggest that the area of suitable physical habitat for adult brown trout increased following the 2013/14 floods and 23% more of this habitat was classified as highly suitable. Additionally, the spatial metrics indicate this physical habitat was more coherent with fewer but larger patches than observed previously (Table 6.4).

The most notable changes in physical habitat provision identified within the results were for the spawning life stage of brown trout. The results suggest that the reach may have provided nearly 250 % more physical habitat for this life stage immediately following the construction of the riffle feature. Over 80 % of this physical habitat was classifiable as highly suitable (Table 6.3) and the results suggest that majority of this additional physical habitat was gained over the riffle feature (Fig. 6.4). It is conceivable that this increase in suitability for spawning resulted from a decrease in water depth over the riffle feature. Interestingly, the physical habitat for this life stage was least suitable over the constructed riffle crest, this may be because the velocities observed within this area were near the upper tolerance for this life stage (Armitage and Ladle, 1991). Further improvements to this habitat were suggested following the morphological changes to the reach during 2013/14 flood events. It is possible that the increased velocities over the riffle feature contributed to a 34 % increase in suitable physical habitat within the reach. Furthermore, 93 % of this habitat was classifiable as highly suitable and more coherent than the previous low flow surveys (Fig. 6.4 and Table 6.3).

The results indicate that the area of suitable physical habitat for fry increased by 74 % immediately following restoration but that it was also still in low abundance (Fig. 6.5). This shallow and slow flowing environment was confined to the margins of the channel and the spatial metrics indicated a high degree of fragmentation (Fig. 6.5 and Table 6.3). However, the percentage of this physical habitat classified as moderate and highly suitable both increased by 21% (Table 6.3). The results suggest after 12 months post construction, physical habitat for fry had the highest proportional improvement in area compared to other life stages, with an increase of ~150 % and ~100 % on the pre-restoration and post-construction surveys, respectively (Table 6.3). However, this habitat was still confined to the channel margins and exhibited a greater number of divisions than any of the previous low flow surveys (Fig 6.4). The overall proportion of fry physical habitat classified highly suitable decreased, but increased in terms of the amount of suitable area (Table 6.3).

Overall, the predicted juvenile brown trout habitat availability over the reach also increased immediately following restoration, particularly in the margins over the riffle feature (Fig. 6.6). Similarly to the spawning and adult physical habitat simulations, the constructed riffle crest was also suggested to be less suitable for juvenile brown trout than the rest of the feature. It is possible that this increase in suitability may have resulted from an increase in water depth as flow was backed-up behind the constructed riffle crest. The results indicate that the proportion of moderate physical habitat suitability increased, however, this was mirrored by slight decreases in both the predicted low and high suitability habitats (Table 6.3). The high suitability habitat still appeared very coherent and accounted for more than 75 % of the total suitable habitat simulated.

The results suggest that the abundance of physical habitat suitable for juveniles decreased slightly following the 2013/14 floods on the immediate post-construction surveys. A loss of suitable habitat was particularly noticeable in both the upstream and downstream pools (Fig. 6.6). This may be a result of an increased water depth in these features resulting from significant scour observed following the 2013/14 floods (Chapter 5). More coherent and highly suitable habitat was observed over the riffle feature. However, scour downstream of the feature may have contributed to some highly suitable habitat being replaced by moderate quality habitat (Table 6.3 & Fig. 6.6).

Post-restoration (moderate and high flows)

Physical habitat abundance and coherence at moderate flows exhibited different patterns to those observed at low flows both pre- and post-flood. The simulations using the 18-month post-construction survey data (post-flood), indicated that a similar amount of physical habitat was available to that afforded by the 3-month post-construction survey (pre-flood) for all life stages. However, the quality of these spaces for the different life stages exhibited some variation.

The results of the simulations using the 3-month post-construction survey data (pre-flood) indicated a significant increase in abundance of suitable physical habitat for adult brown trout over the riffle feature compared to low flows immediately post-construction (Fig .6.3). The total habitat was classed as either high or low habitat suitability, with these classes accounting for ~43 % and ~57 % respectively. The former dominated over the riffle feature, possibly due to a preferential increase in velocity and depth for this life stage. Low suitability physical habitat was observed abundantly upstream and downstream of the riffle feature in deeper areas of the channel which may have been less suitable (Fig. 6.3). The spatial metrics indicated that these spaces were reasonably coherent (Table 6.3).

The results suggest that following the 2013/14 floods physical habitat suitable for adult brown trout habitat decreased by 8% in comparison to the pre-flood moderate flow. However, the coherence of these spaces was maintained and the abundance of highly suitable physical habitat increased. Highly suitable habitat was simulated to be a marginally larger area in total than the less suitable physical habitat. In comparison to the post-flood low flow (12-month post-construction survey), the simulation indicated that the abundance of physical habitat suitable for adult brown trout significantly increased in the moderate flow following the 2013/14 floods (Table 6.3). Therefore, this pattern reflects a similar relationship for adult brown trout between low and moderate flows to that observed prior to the 2013/14 floods. Significantly more suitable habitat was observed over the riffle feature, particularly over the head, as a possible result of an increase in water depth to within the suitable range (Fig. 6.3). The highly suitable habitat accounted for proportionally less of the total habitat than observed during the post-flood low flow, but the spatial metrics indicated it was more coherent (Table 6.3).

There were no comparable near bankfull flows for the physical habitat simulations using the 7-month post-construction survey data. However, given the similar post-flood morphology, broad comparisons can be made to the post-flood low and moderate surveys. The results suggest that the physical habitat suitable for adults was most abundantly observed during this survey. This was not totally surprising given the increase in wetted channel area (Table 6.3). However, 84 % of the habitat was classified as low suitability, and the upstream and downstream pools (previously observed at moderate flows to be suitable habitat) were observed in this high flow as unsuitable habitat (Fig. 6.3). These observations may largely be due to a significant increase in water depth throughout the reach to the upper bounds of the depth suitability criteria for this life stage (Armitage and Ladle, 1991). The spatial metrics indicated these habitat spaces were more coherent at high flows than at low flows, yet less coherent than at moderate flows (Table 6.3).

The results suggest that 21 % more spawning habitat for brown trout was available during the moderate flow 3 months post-construction than the low flow captured immediately post-construction. However, this physical habitat was generally of a lower suitability than observed at the lower flows possibly due to an increase in water depth which may have afforded a less preferable environment. A similar spatial configuration of this physical habitat was also simulated for the moderate flow observed following the 2013/14 flood events (18-months post-construction survey). Interestingly, the physical habitat near post-flood crest (head of the riffle feature) was more suitable than that suggested over the rest of the feature. It is possible that the deposition in this area during the 2013/14 floods may have afforded a shallower environment that was preferable for spawning and contributed to an overall 21 % increase in suitable physical habitat for this life stage.

A very similar relationship of the physical habitat for spawning brown trout was observed between the low and moderate flows following the 2013/14 flood events to that observed previously (i.e. in greater abundance but of a lower suitability). This relationship was not maintained at higher flows. As water depth increased, the abundance of this physical habitat significantly reduced and became restricted to the shallower parts of the channel (i.e. the channel margins and the post-flood riffle crest). The results also suggest that the spawning physical habitat became very fragmented during this high flow.

Physical habitats suitable for fry were confined to the shallower slower flowing areas of the channel in the margins at moderate flows prior to the 2013/14 flood events (Fig. 6.5). The results suggest that less of this physical habitat was available despite an increase in the wetted channel area and that 95% of these spaces were classified as either low or moderate suitability for this life stage (Table 6.3). This was a 35 % increase on the low flow captured immediately following the construction of the riffle feature. Additionally, the spatial metrics indicated that physical habitat suitable for this life stage became more fragmented as discharge increased (Table 6.3).

Following the 2013/14 flood events, the results suggest that shallow, slow flowing areas of the channel suitable for fry were mostly retained at moderate flows. These spaces remained fragmented and were still not abundant throughout the reach. The relationship of this habitat between low and moderate flows was similar to that observed prior to the 2013/14 floods in that the area of suitable physical habitat significantly decreased (-79 %). However, the results indicate that the area of suitable physical habitat was nearly twice as abundant at high flows than at moderate flows, but still less abundant than observed at low flows. It is possible that as the complex bank morphology within the reach (i.e. steps) may have afforded additional shallow, slow flowing physical habitat suitable for fry as they became submerged.

The simulation results suggest that physical habitat suitable for juvenile brown trout was more abundant at moderate flows than low flows prior to the 2013/14 flood events. Highly suitable physical habitat accounted for a small proportion (21 %) of total available physical habitat than observed at low flows and was confined to the shallower areas in the channel margins (Fig. 6.6 and Table 6.3). Moderately suitable physical habitat was the dominant class of habitat (72 %) and was particularly noted over the riffle feature (Fig. 6.6). The areas of unsuitable habitat were observed in the deeper areas of the channel, immediately downstream of the riffle feature and in the downstream pool where scour was observed.

The results suggest physical habitat suitable for juvenile brown trout was also more abundant at moderate flows than low flows following the 2013/14 flood events. In addition, highly suitable physical habitats increased by 11 % on the pre-flood moderate flow and were less fragmented. However, the physical habitat suitability for this life stage decreased in the newly formed upstream pool probably due to a decrease in water depth. The physical habitat suitable for juveniles was retained in a high abundance during the high flows but it was restricted to the shallower areas of the channel (i.e. the riffle feature and the channel margins) (Fig. 6.6). The proportion of highly suitable physical habitat decreased as depth increased with discharge (Table 6.3 and Fig. 6.6). These results indicate that depth may have been a primary control on the suitability of physical habitat for this life stage.

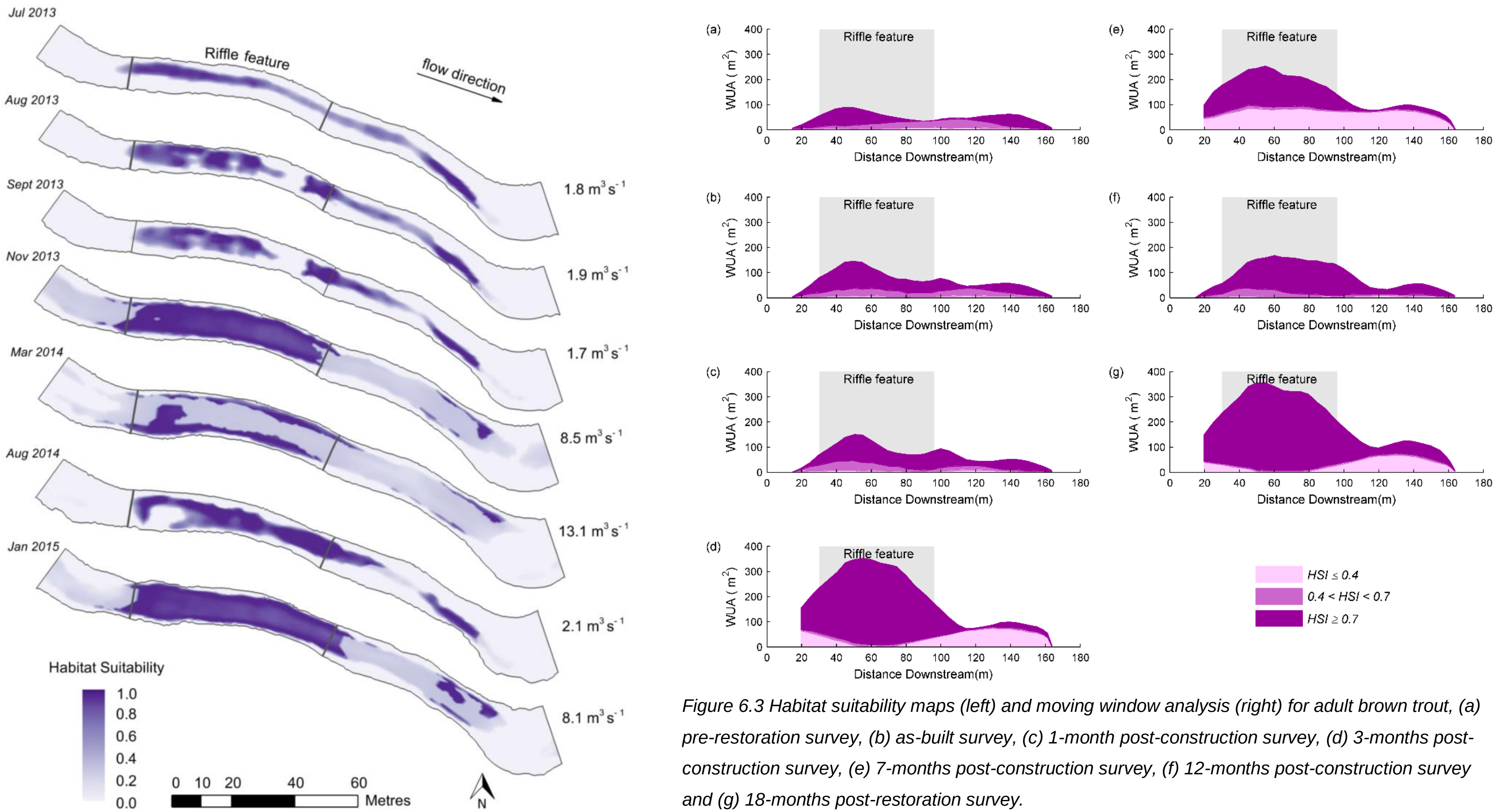


Figure 6.3 Habitat suitability maps (left) and moving window analysis (right) for adult brown trout, (a) pre-restoration survey, (b) as-built survey, (c) 1-month post-construction survey, (d) 3-months post-construction survey, (e) 7-months post-construction survey, (f) 12-months post-construction survey and (g) 18-months post-restoration survey.

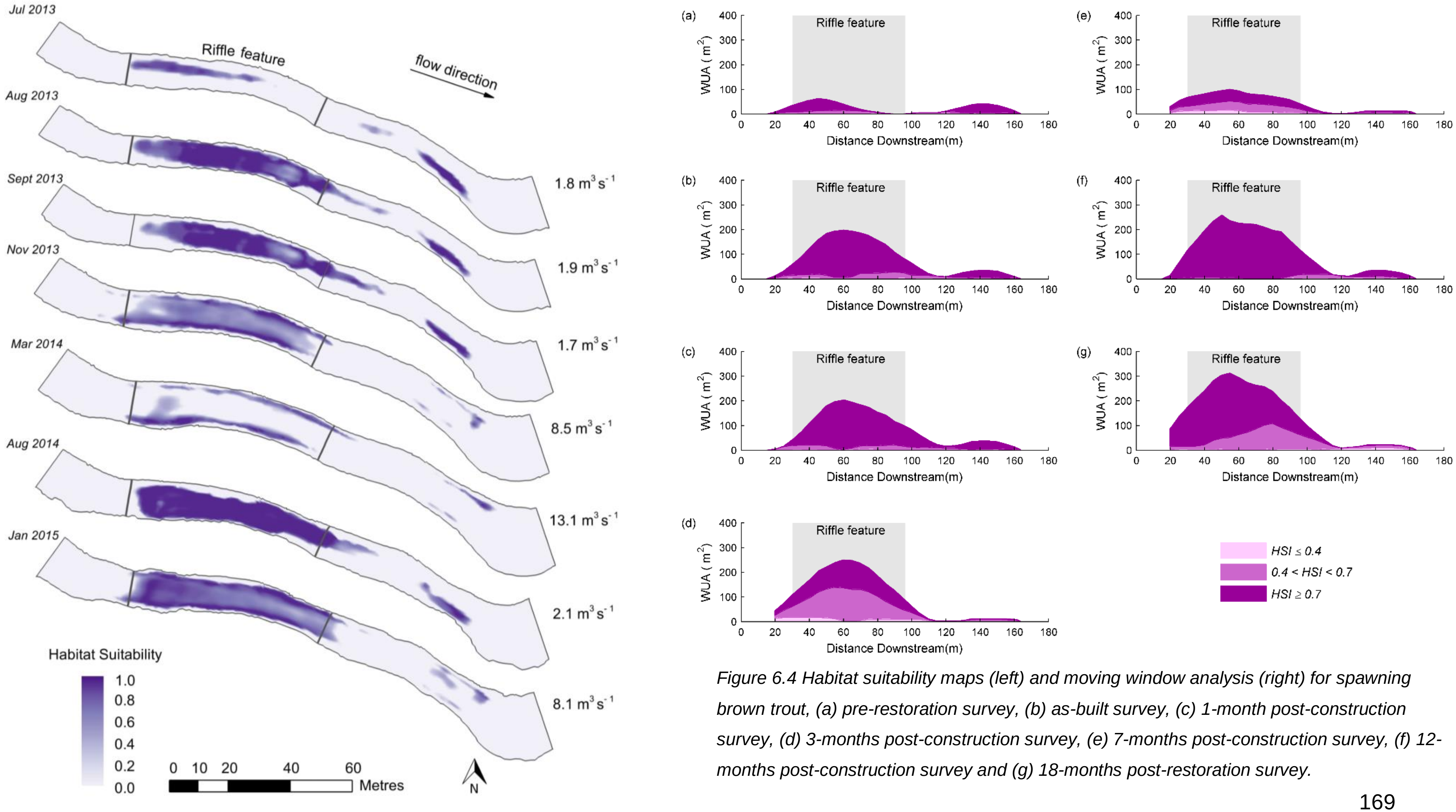
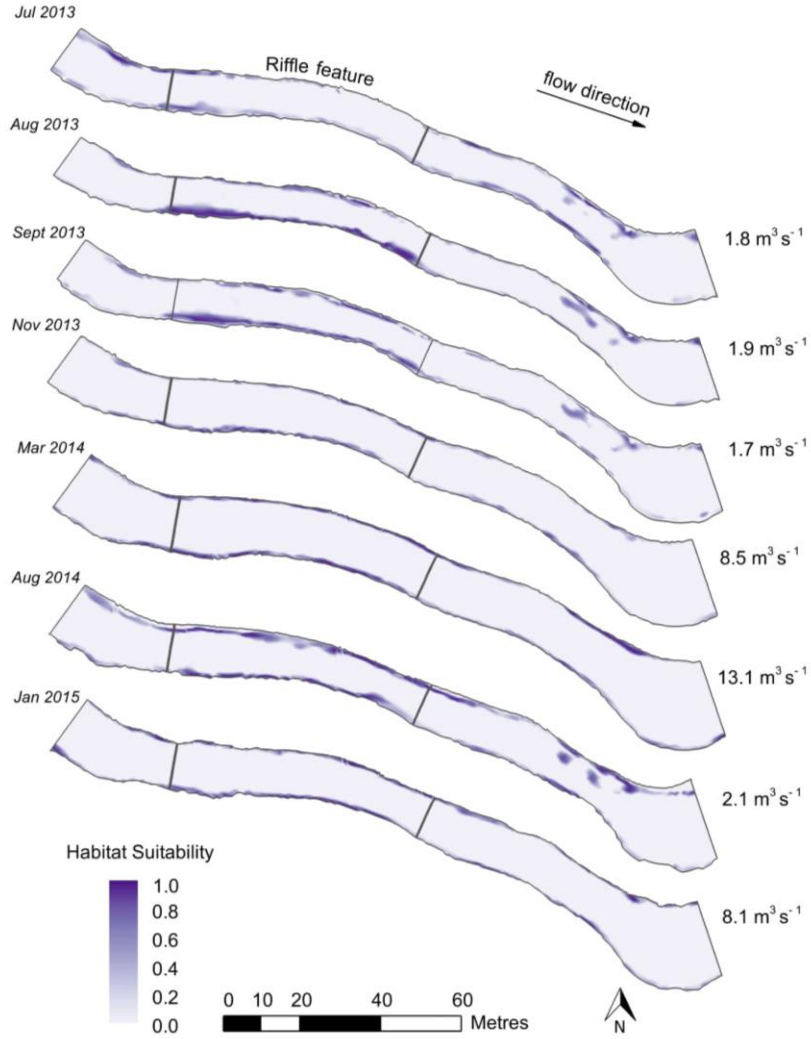


Figure 6.4 Habitat suitability maps (left) and moving window analysis (right) for spawning brown trout, (a) pre-restoration survey, (b) as-built survey, (c) 1-month post-construction survey, (d) 3-months post-construction survey, (e) 7-months post-construction survey, (f) 12-months post-construction survey and (g) 18-months post-restoration survey.



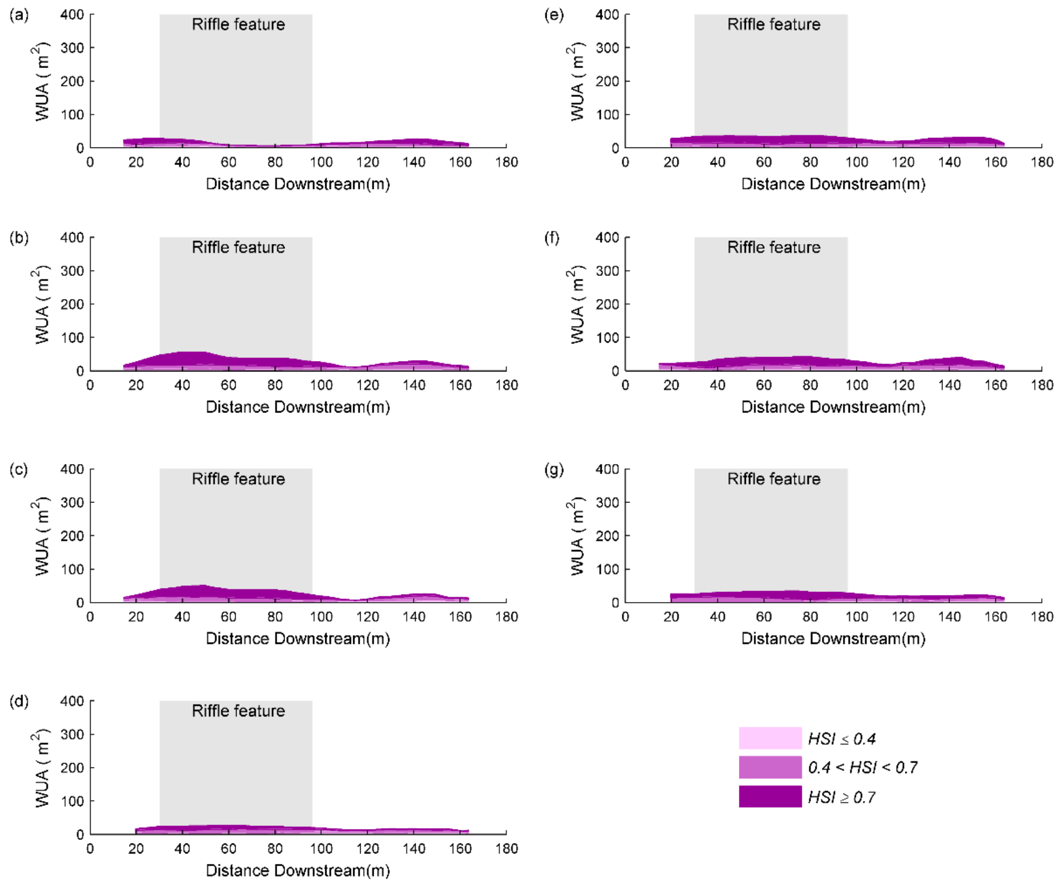


Figure 6.5 Habitat suitability maps (left) and moving window analysis (right) for fry brown trout, (a) pre-restoration survey, (b) as-built survey, (c) 1-month post-construction survey, (d) 3-months post-construction survey, (e) 7-months post-construction survey, (f) 12-months post-construction survey and (g) 18-months post-restoration survey.

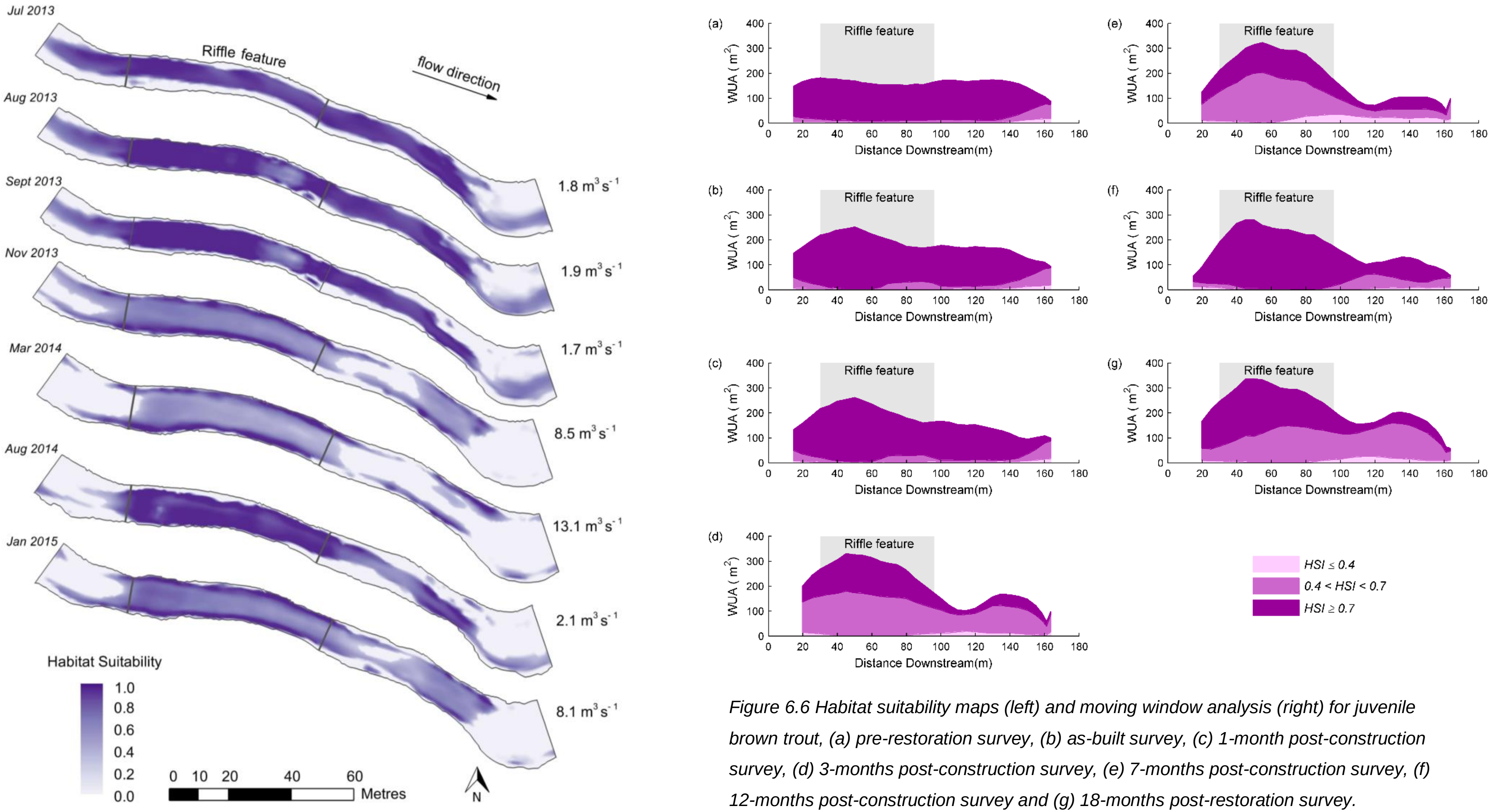


Figure 6.6 Habitat suitability maps (left) and moving window analysis (right) for juvenile brown trout, (a) pre-restoration survey, (b) as-built survey, (c) 1-month post-construction survey, (d) 3-months post-construction survey, (e) 7-months post-construction survey, (f) 12-months post-construction survey and (g) 18-months post-restoration survey.

Table 6.3 Spatial metrics of habitat suitability for brown trout

		Habitat Suitability											
		Low				Moderate				High			
	Total Area (m^2)	% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension
Adult Brown trout													
Baseline	318	27	6	1.51	1.56	37	5	1.45	1.57	36	3	1.26	1.51
As-built	423	19	11	2.18	1.72	30	10	1.81	1.67	51	5	0.86	1.43
1 Month	395	15	10	2.22	1.71	27	7	1.74	1.64	58	4	0.88	1.44
3 Months	1319	57	3	0.38	1.29	-	-	-	-	43	3	0.34	1.22
7 Months	1405	84	4	0.38	1.33	-	-	-	-	16	5	1.26	1.57
12 Months	457	17	8	1.84	1.64	9	6	2.19	1.67	74	2	0.66	1.38
18 Months	1213	47	3	0.46	1.32	-	-	-	-	53	2	0.35	1.25
Juvenile Brown trout													
Baseline	925	8	4	1.25	1.46	9	8	1.43	1.54	82	1	0.41	1.32
As-built	1077	5	7	1.97	1.65	18	7	1.16	1.53	77	1	0.40	1.31
1 Month	998	5	5	2.12	1.67	17	11	1.04	1.48	78	1	0.43	1.33
3 Months	1215	7	17	2.19	1.73	72	3	0.54	1.41	21	8	1.59	1.67
7 Months	1176	18	23	1.71	1.68	62	10	0.64	1.45	20	7	1.70	1.69
12 Months	967	7	9	2.13	1.70	17	11	1.18	1.52	76	4	0.35	1.26
18 Months	1264	10	14	1.76	1.66	58	6	0.62	1.43	32	6	1.04	1.55

Table 6.3 continued

		Habitat Suitability											
		Low				Moderate				High			
	Total Area (m^2)	% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension
Fry Brown trout													
Baseline	58	74	16	3.24	1.89	17	10	5.16	2.22	9	4	5.71	2.44
As-built	101	32	11	3.60	1.94	38	12	2.81	1.81	30	7	3.28	1.88
1 Month	93	42	11	2.96	1.84	39	13	3.11	1.86	20	10	4.02	2.00
3 Months	47	55	17	5.75	2.22	40	18	6.35	2.31	4	6	12.86	5.38
7 Months	72	36	15	5.93	2.24	36	23	6.77	2.32	28	18	8.44	2.50
12 Months	198	51	24	2.80	1.85	32	16	2.73	1.82	17	10	4.13	2.02
18 Months	42	32	9	5.19	2.20	50	16	6.47	2.32	18	14	8.83	2.77
Spawning Brown trout													
Baseline	115	15	5	3.23	1.85	28	6	2.26	1.67	57	3	1.51	1.53
As-built	399	3	5	4.05	2.01	13	6	1.92	1.63	84	2	0.59	1.34
1 Month	425	2	4	3.27	1.82	16	6	1.58	1.56	82	2	0.59	1.35
3 Months	515	16	6	1.09	1.41	58	5	0.68	1.38	26	7	1.67	1.64
7 Months	205	35	12	2.22	1.72	38	11	2.46	1.78	27	6	2.83	1.83
12 Months	534	3	5	2.43	1.64	4	3	2.11	1.58	93	2	0.35	1.22
18 Months	627	10	8	1.81	1.62	32	4	0.89	1.43	58	2	0.67	1.39

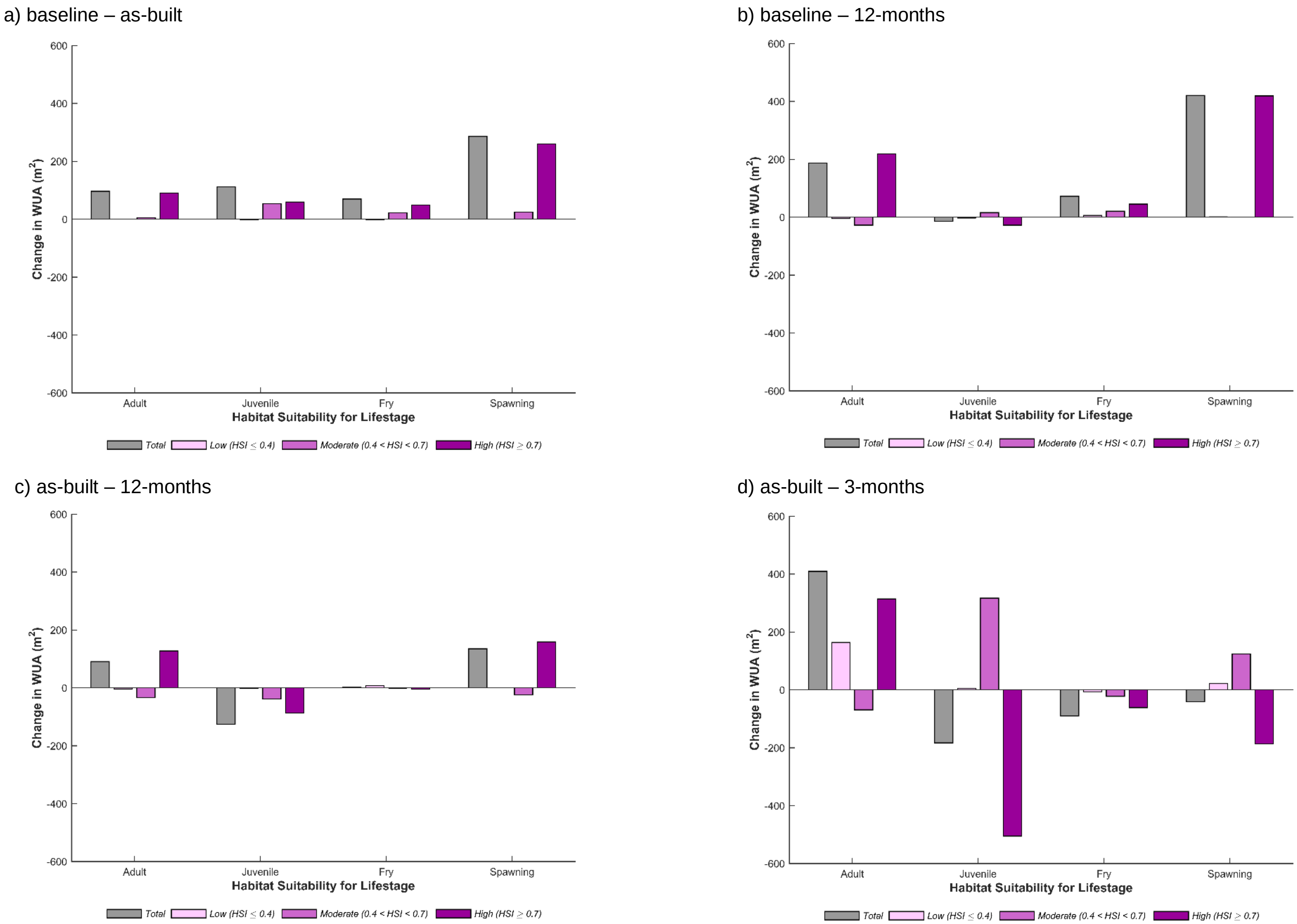
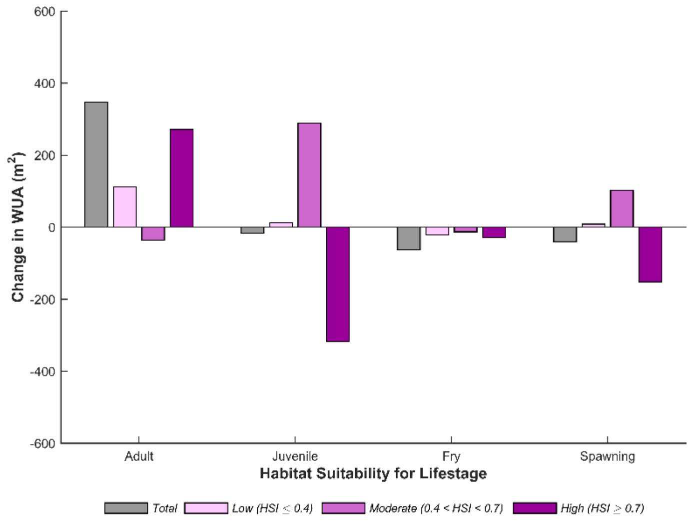
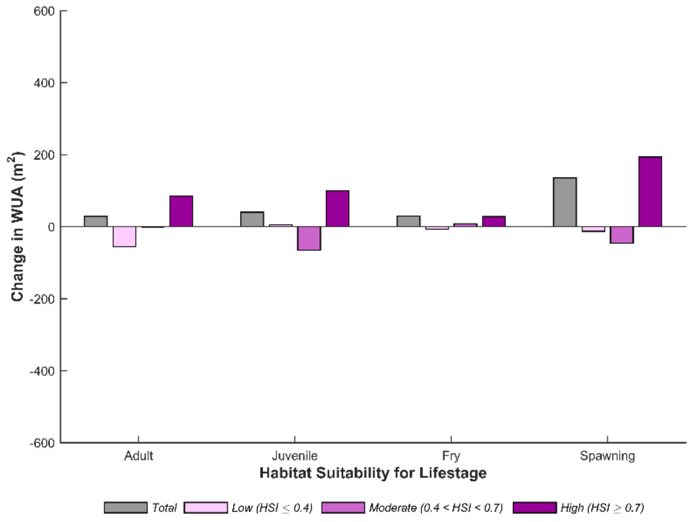


Figure 6.7 Change in WUA for brown trout between surveys a) baseline and as-built, b) baseline and 12-months, c) as-built and 12-months d) as-built and 3-months e) 12-months to 18-months, f) 3-months to 18-months g) 7-months to 18-months.

e) 12-motnhs – 18-months



f) 3-motnhs – 18-months



g) 7-motnhs – 18-months

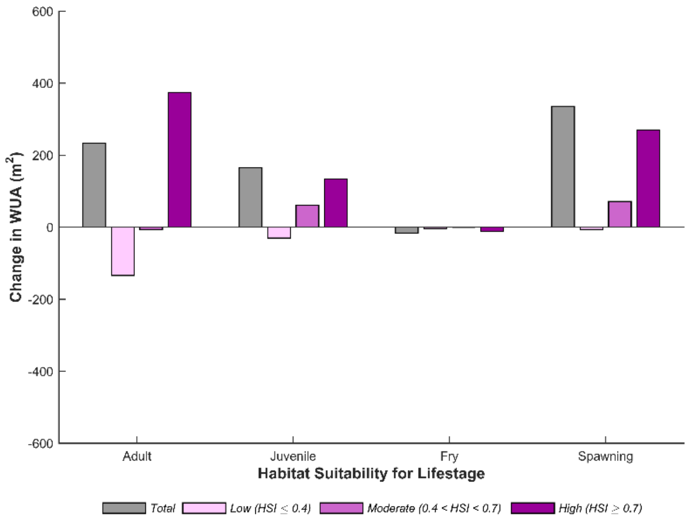


Figure 6.7 (continued) Change in WUA for brown trout between surveys a) baseline and as-built, b) baseline and 12-months, c) as-built and 12-months d) as-built and 3-months e) 12 months to 18-months, f) 3-months to 18-months g) 7-months to 18-months.

6.3.2 Dace

Pre-restoration condition (low flow)

Physical habitat for dace, a species recorded within the lower catchment prior to restoration, exhibited similar trends to the brown trout physical habitat. Prior to restoration, the simulation results indicated that adult dace physical habitat (which prefer depths between 0.4 and 1 m and velocities between 0.2 and 0.7 m s^{-1} (Armitage and Ladle, 1991)) was highly abundant, suitable and coherent along the reach (Fig. 6.8). In total, 52% of the total suitable physical habitat was classified as highly suitable and located within the deeper and marginally faster flowing central areas of the channel (Table 6.4). The downstream pool, however, presented largely low and moderately suitable physical habitat as it may have been too deep and slow flowing to be a preferential physical habitat for adult dace prior to restoration (Chapter 5).

Physical habitat suitable for spawning dace was in a very low abundance, with areas too small to calculate spatial metrics (Fig. 6.11). This life stage prefers depths between 0.2 and 0.8 m which were adequately provisioned within the reach (Armitage and Ladle, 1991). However, this life stage prefers velocities between 0.5 and 1 m s^{-1} which were found only in a small area in the reach (Armitage and Ladle, 1991). Therefore, this would suggest that a lack of diversity in velocities within the reach limited the physical habitat performance for spawning dace prior to restoration. The results also suggest that physical habitat for fry (which prefer depths up to 0.3 m and velocities up to 0.25 m s^{-1} (Armitage and Ladle, 1991)) was also in a low abundance, largely low suitability and fragmented prior to restoration as it was confined to the shallower channel margins (Fig. 6.10).

Conversely, the results indicate that the reach may have afforded abundant habitat suitable for juvenile dace (Fig. 6.9). This life stage prefers depths between 0.3 and 0.8 m and slower velocities between 0.1 and 0. 4 m s^{-1} (Armitage and Ladle, 1991). Whilst this habitat was widely available throughout the reach, it was less suitable than for adult dace as adults may tolerate slightly deeper environments (Table 6.4). Therefore, the downstream pool was likely to have been unsuitable for juvenile dace prior to restoration as it was too deep.

Post-restoration (low flows)

Immediately following restoration, the abundance of total physical habitat suitable for adult dace was maintained, but spatially reconfigured at low flows. The simulations suggest that the riffle crest became a less suitable habitat (Fig. 6.8) potentially because the elevated velocities (Chapter 5) were out of the preferential range for this life stage of dace. Over the rest of the riffle feature, the physical habitat became more suitable as the water depths were reduced and velocities increased slightly. The simulated physical habitat using the 12-month post-construction survey data indicated a considerable re-working of physical habitat following the 2013/14 floods. The total provision of physical habitat suitable for adult dace habitat increased by 29%, with the riffle feature still affording highly suitable physical habitat. However, physical habitat suitability decreased where scour processes had increased the water depth. Interestingly, physical habitat for adult dace was also less suitable in the areas of higher velocity over the riffle feature, particularly over the riffle crest.

On construction, the results suggest that the riffle feature created nearly 300 m^3 of physical habitat highly suitable for spawning (Fig. 6.11 and Table 6.4) through elevating velocities above 0.5 m s^{-1} and reducing water depth to less than 0.8 m. Interestingly, small areas of low suitability coincided with the peak velocities over the riffle crest. Following the significant morphological adjustments within the reach during the 2013/14 floods, the area of highly suitable physical habitat further increased by approximately 30%. It is estimated that 98% of this physical habitat would have been highly suitable and coherent for this life stage (Table 6.4).

The results suggest that immediately following restoration the area suitable for fry more than tripled. However, this physical habitat was still found in low abundance in the channel margins and was highly fragmented (Table 6.4 & Fig. 6.10). Physical habitat suitable for fry was predicted in the greatest abundance within whole monitoring period during the 12-month post-construction survey. However, this was still in a relatively low abundance and highly fragmented when compared with the habitat provision for other life stages.

Following construction of the riffle feature, the area of physical habitat suitable for juvenile dace was mostly retained. However, the results indicate that an increase in velocities over the constructed riffle crest above 0.7 m s^{-1} may have significantly reduced the suitability of this area for juvenile dace (Fig. 6.9). The abundance of this physical habitat suitable for juvenile dace habitat was maintained at low flows following the 2013/14 floods. The quality of this habitat was significantly reduced when compared to the pre -and immediate post-restoration surveys (Table 6.4 & Fig. 6.9). The dominant class of physical habitat was low suitability (53 %) with only 10% of the habitat classed as highly suitable. This small amount of highly suitable habitat was also observed to be highly fragmented by higher patches of velocity (Table 6.4).

Post-restoration (moderate and high flows)

The simulations suggest that prior to the 2013/14 floods, the available physical habitat suitable for adult dace increased by 60% throughout the reach at moderate flows compared to low flows, as the water depth increased in the channel margins. This was particularly noticeable upstream of the riffle crest (Fig. 6.8). The physical habitat was less suitable over the riffle feature than at lower flows, possibly due to the higher velocities experienced in this area. However, the overall suitability of this physical habitat for adult dace was lower with 81% of habitat classified as either moderate or low suitability. Following the morphological adjustments to the reach during the 2013/14 floods, the results suggested a very similar pattern at moderate flows. A slight increase in moderately suitable physical habitat was observed as the post-flood riffle crest extended the area of higher velocities over the riffle feature (Fig. 6.8). At higher flows the abundance of suitable physical habitat increased with wetted area. Moreover, as the influence of the riffle crest on velocity drowned out with an increase in discharge, the physical habitat suitability for adult dace improved in this area of the channel.

The results suggest that the area of physical habitat suitable for spawning dace prior to the 2013/14 flood events significantly decreased at moderate flows. Very small areas in low abundance were available and were likely to have been available in the channel margins as water depths increased with discharge (Fig. 6.11). This habitat was also generally of a poor suitability and highly fragmented. Following the 2013/14 floods, the provision of suitable spawning physical habitat was significantly improved at moderate flows. This physical habitat was more coherent and largely of a higher suitability, but when compared to other life stages, it was still simulated in a relatively low abundance. The results also suggest it would have been restricted to shallower areas of the channel such as the margins and the post-flood riffle crest (Fig. 6.11). At higher flows following the flood events, the physical habitat suitability was reduced and further confined and fragmented along the channel margins (Fig. 6.11 and Table 6.4).

The results indicate that the physical habitat provision for fry was negligible during all moderate flows captured in the monitoring period. Where this physical habitat was afforded, it was highly fragmented into shallow areas of the channel margins (Fig. 6.10 and Table 6.4). The simulations suggest that suitable physical habitat may have increased at higher flows following the 2013/14 flood events as additional shallow, slow flowing habitat became available with the submergence of the complex bank morphology.

Both before and after the 2013/14 flood events physical habitat suitable for juvenile dace habitat saw a significant decrease in total available habitat at moderate flows and high flows when compared to low flows. In both simulations, the physical habitat became confined to a narrow corridor of along the channel margins as the channel became too deep for this life stage. In all of these moderate and high flows surveys, ~70 % of the physical habitat afforded by the reach classified as low suitability.

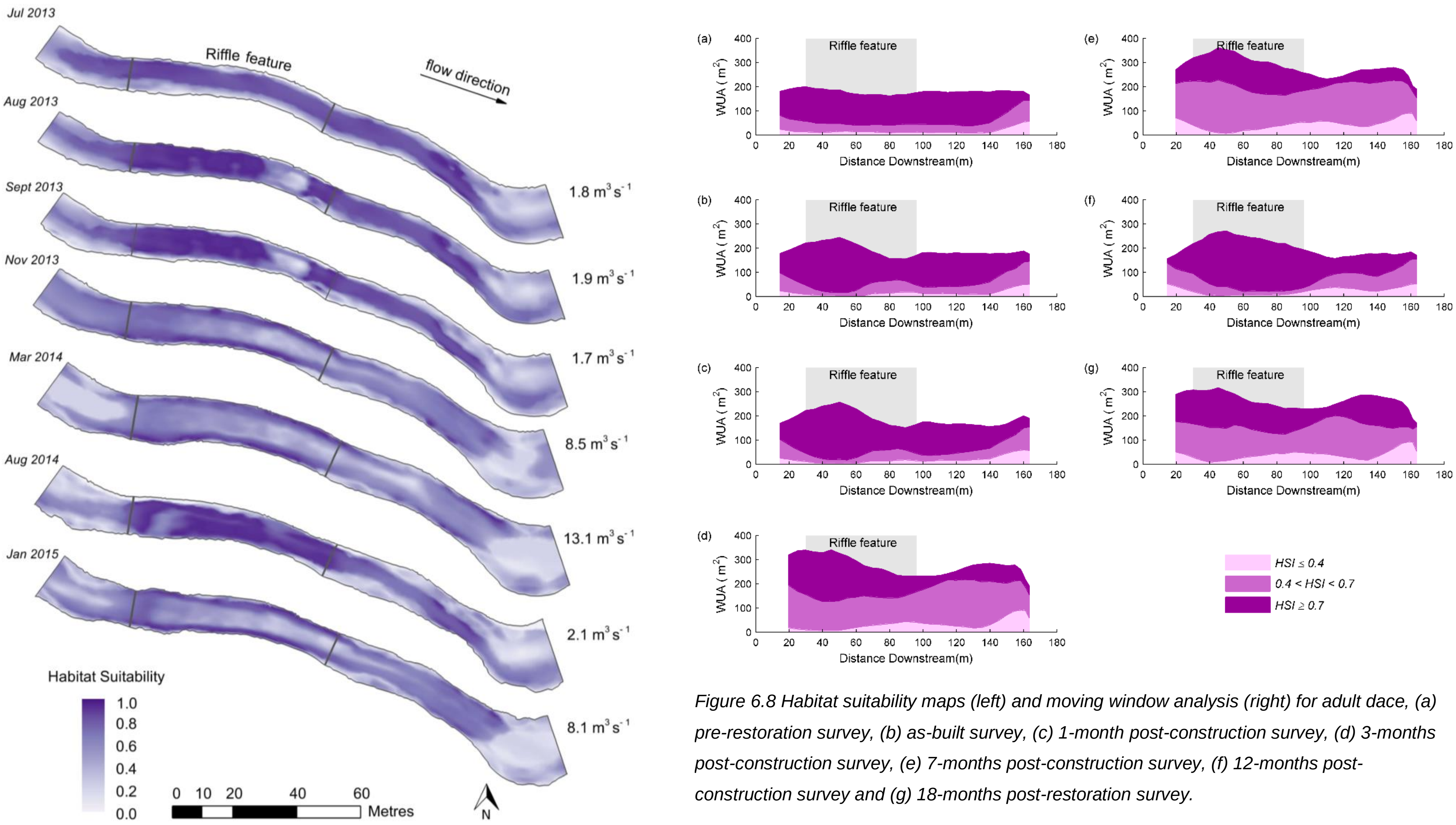
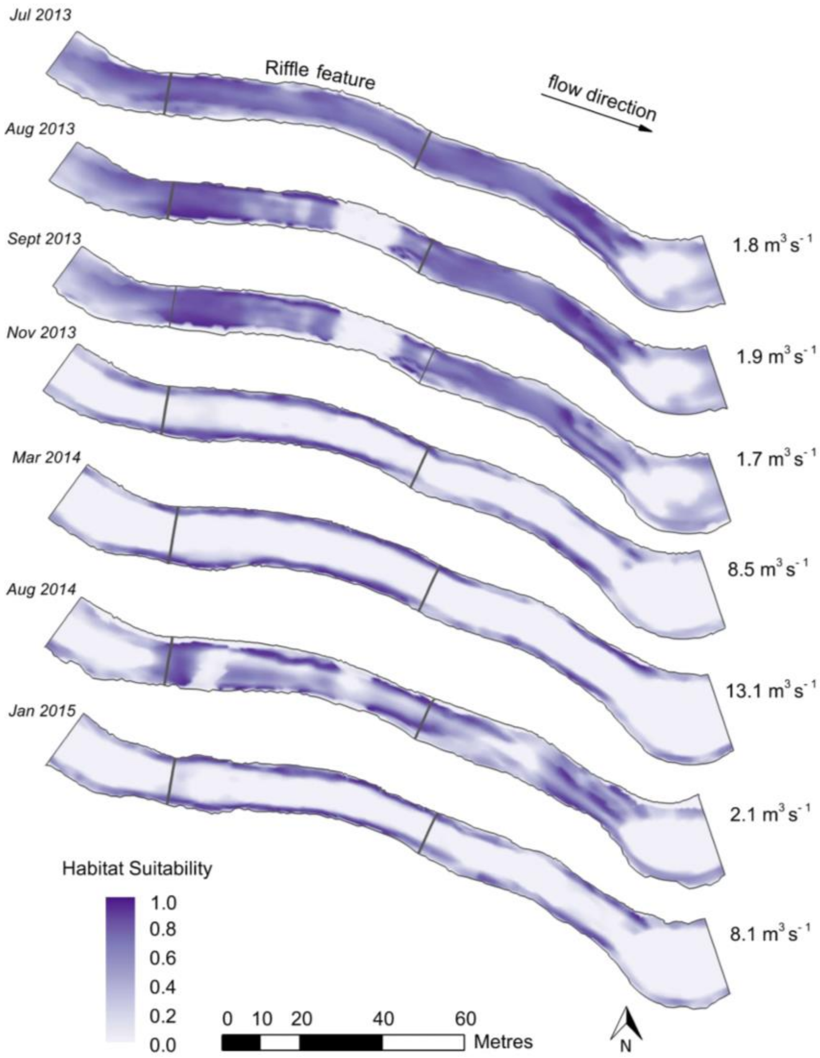


Figure 6.8 Habitat suitability maps (left) and moving window analysis (right) for adult dace, (a) pre-restoration survey, (b) as-built survey, (c) 1-month post-construction survey, (d) 3-months post-construction survey, (e) 7-months post-construction survey, (f) 12-months post-construction survey and (g) 18-months post-restoration survey.



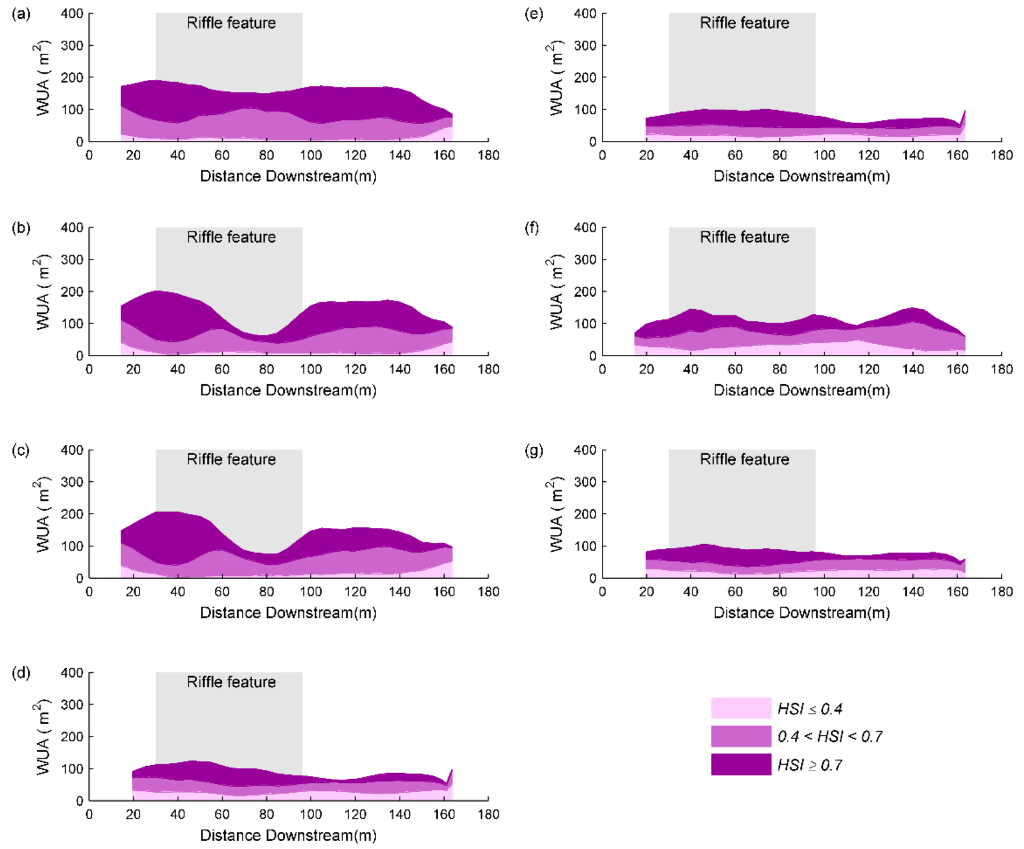
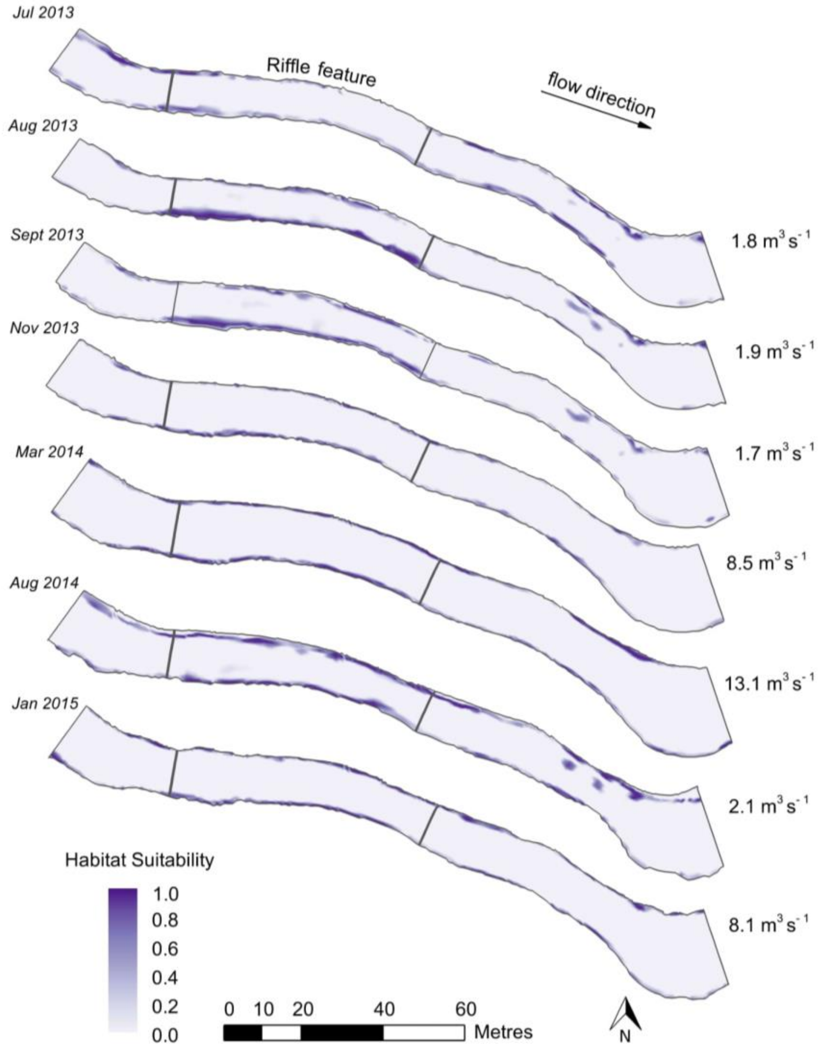


Figure 6.9 Habitat suitability maps (left) and moving window analysis (right) for juvenile dace, (a) pre-restoration survey, (b) as-built survey, (c) 1-month post-construction survey, (d) 3-months post-construction survey, (e) 7-months post-construction survey, (f) 12-months post-construction survey and (g) 18-months post-restoration survey.



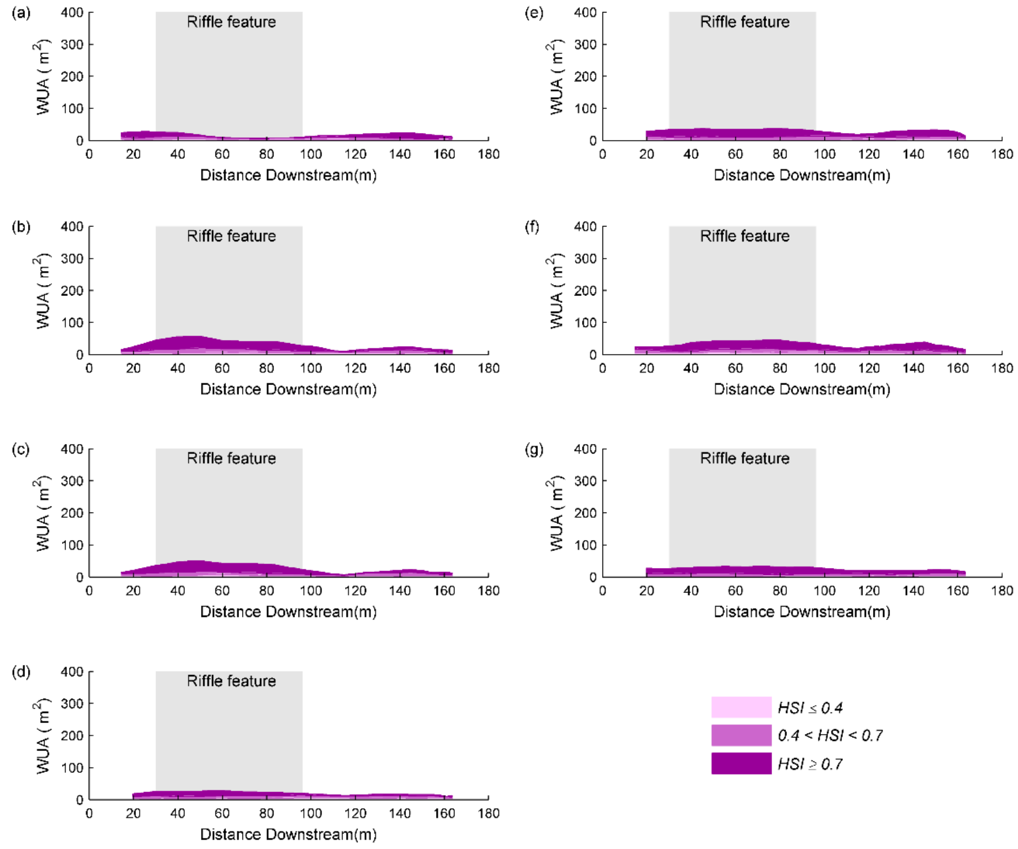


Figure 6.10 Habitat suitability maps (left) and moving window analysis (right) for fry dace, (a) pre-restoration survey, (b) as-built survey, (c) 1-month post-construction survey, (d) 3-months post-construction survey, (e) 7-months post-construction survey, (f) 12-months post-construction survey and (g) 18-months post-restoration survey.

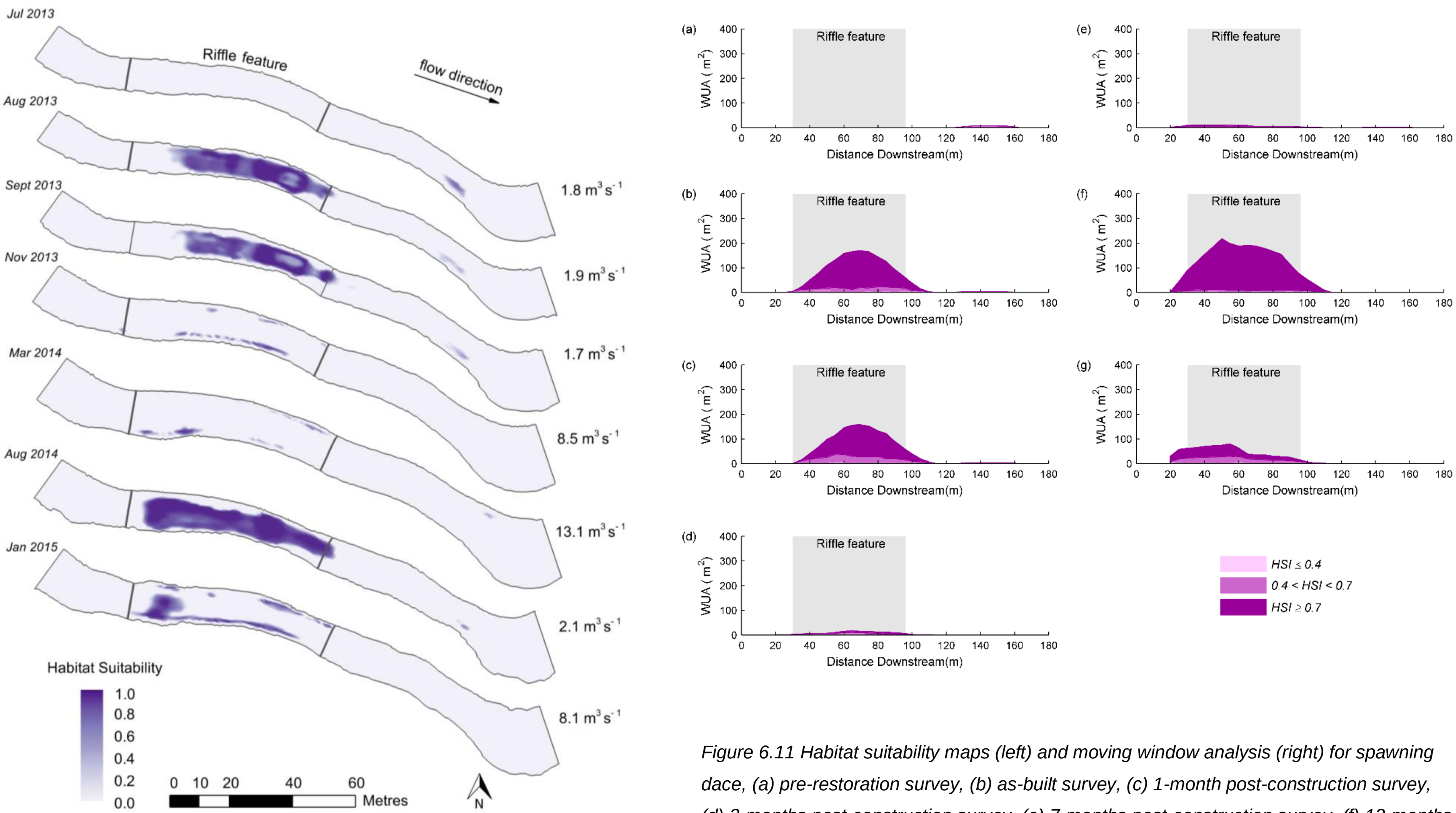


Figure 6.11 Habitat suitability maps (left) and moving window analysis (right) for spawning dace, (a) pre-restoration survey, (b) as-built survey, (c) 1-month post-construction survey, (d) 3-months post-construction survey, (e) 7-months post-construction survey, (f) 12-months post-construction survey and (g) 18-months post-restoration survey.

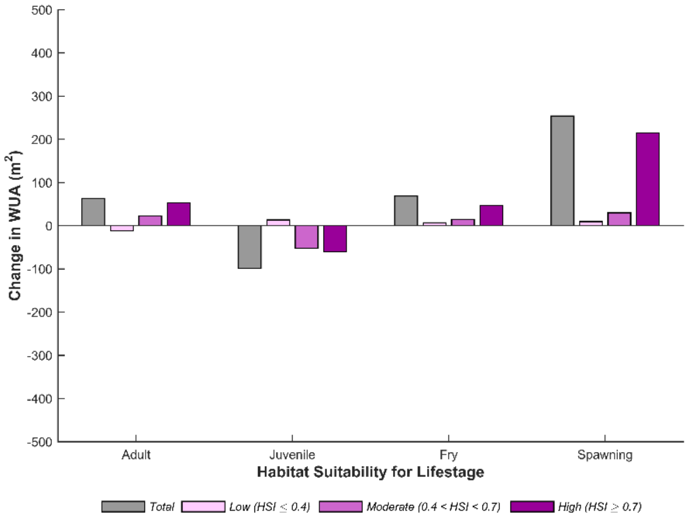
Table 6.4 Spatial metrics of habitat suitability for dace

		Habitat Suitability											
		Low				Moderate				High			
	Total Area (m^2)	% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension
Adult Dace													
Baseline	1338	23	14	1.36	1.62	25	22	1.60	1.69	52	3	0.47	1.34
As-built	1384	21	15	1.10	1.54	26	10	1.28	1.61	53	4	0.44	1.33
1 Month	1500	26	12	1.00	1.54	26	13	1.19	1.60	47	3	0.45	1.34
3 Months	2219	24	15	0.89	1.52	57	8	0.40	1.36	19	5	1.03	1.55
7 Months	2526	36	19	0.65	1.47	53	5	0.45	1.39	12	7	1.41	1.63
12 Months	1780	34	8	0.75	1.48	27	13	0.92	1.52	39	3	0.40	1.30
18 Months	2223	34	10	0.75	1.49	47	6	0.63	1.47	20	3	1.03	1.55
Juvenile Dace													
Baseline	1387	17	13	1.31	1.59	44	5	0.83	1.51	39	4	0.61	1.40
As-built	1305	25	11	1.11	1.56	41	11	0.85	1.51	33	7	0.65	1.40
1 Month	1435	26	13	1.15	1.58	45	8	0.65	1.44	30	4	0.73	1.44
3 Months	948	70	14	1.12	1.61	23	16	1.96	1.73	7	14	3.84	1.98
7 Months	689	66	21	1.45	1.67	22	27	2.85	1.87	12	13	4.12	2.01
12 Months	1384	53	8	0.85	1.53	38	15	0.89	1.52	10	10	2.13	1.74
18 Months	772	72	12	1.29	1.64	21	26	2.61	1.83	7	20	4.71	2.08

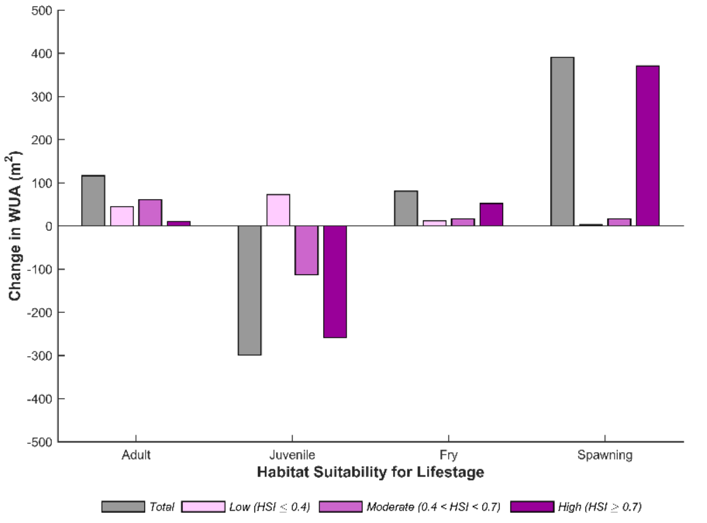
Table 6.4 continued

		Habitat Suitability											
		Low				Moderate				High			
	Total Area (m^2)	% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension
Fry Dace													
Baseline	26	60	10	4.94	2.15	18	9	7.00	2.72	22	5	6.11	2.49
As-built	87	35	12	3.51	1.92	29	10	3.46	1.91	36	12	3.69	1.95
1 Month	80	51	13	3.20	1.88	25	7	2.81	1.77	24	11	5.45	2.21
3 Months	27	57	13	5.86	2.28	38	14	7.57	2.55	5	4	11.05	8.12
7 Months	51	28	7	6.25	2.34	28	13	6.19	2.33	44	16	8.31	2.47
12 Months	168	46	22	3.03	1.87	31	16	3.65	1.95	23	14	4.33	2.04
18 Months	33	30	8	5.08	2.21	39	8	5.55	2.26	31	17	8.86	2.69
Spawning Dace													
Baseline	-	-	-	-	-	-	-	-	-	-	-	-	-
As-built	297	9	3	1.79	1.51	13	3	1.75	1.55	78	1	0.52	1.25
1 Month	273	8	6	2.62	1.72	22	6	1.56	1.54	70	3	0.62	1.29
3 Months	4	52	3	8.44	3.93	-	-	-	-	27	1	5.71	8.25
7 Months	9	40	4	6.03	2.62	43	4	5.20	2.38	17	1	4.51	2.53
12 Months	392	-	-	-	-	2	4	3.65	1.91	98	1	0.36	1.19
18 Months	82	20	2	1.78	1.42	33	9	3.09	1.84	47	4	1.38	1.42

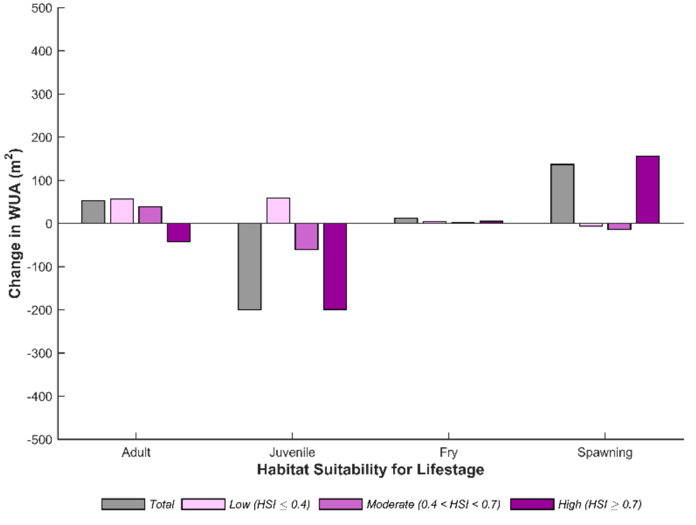
a) baseline – as-built



b) baseline – 12-months



c) as-built – 12-months



d) as-built – 3-months

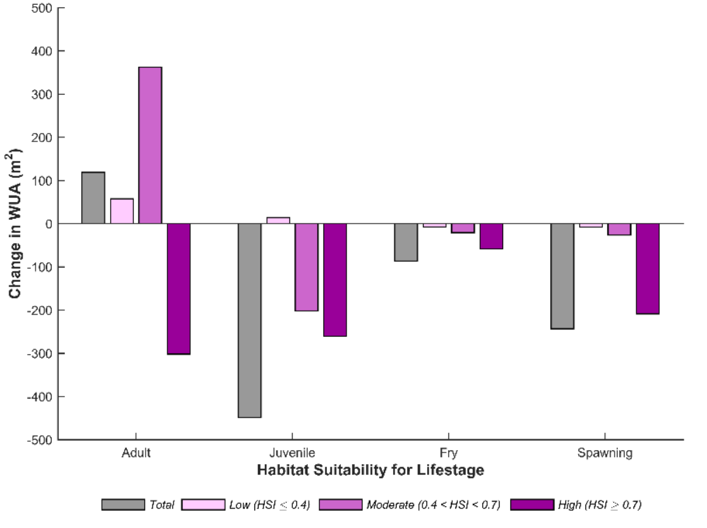
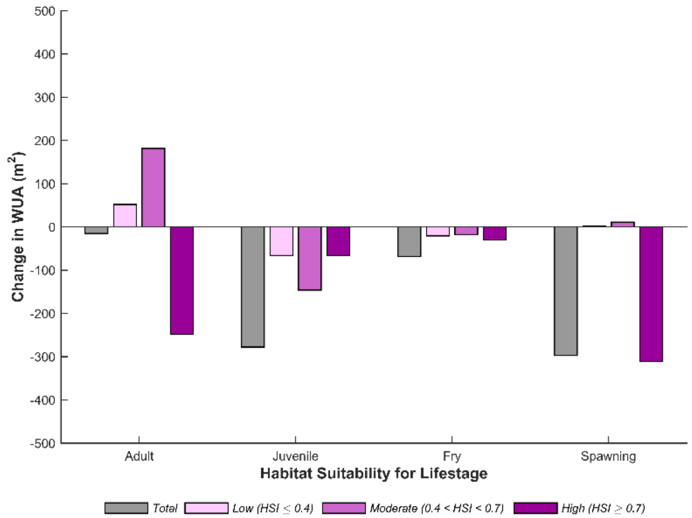
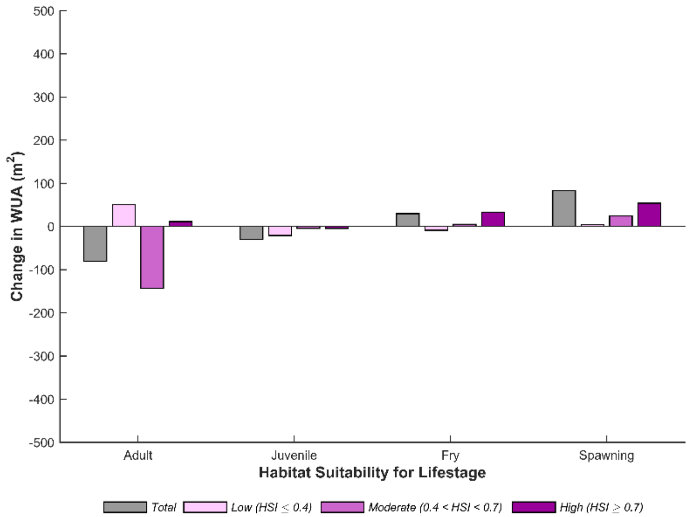


Figure 6.12 Change in WUA for dace between surveys a) baseline and as-built, b) baseline and 12-months, c) as-built and 12-months d) as-built and 3 months e) 12 months to 18 months, f) 3-months to 18-months g) 7-months to 18 months.

e) 12-motnhs – 18-months



f) 3-motnhs – 18-months



g) 7-motnhs – 18-months

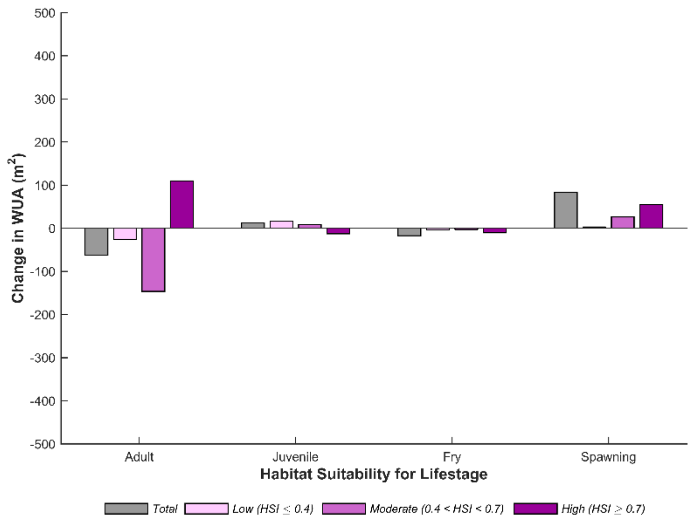


Figure 6.12 (continued) Change in WUA for dace between surveys a) baseline and as-built, b) baseline and 12-months, c) as-built and 12-months d) as-built and 3-months e) 12-months to 18-months, f) 3-months to 18-months g) 7-months to 18-months.

6.3.3 Roach

Pre-restoration condition (low flow)

Roach was also observed in the lower catchment prior to the construction of the riffle feature. The habitat preference curves for the adult and juvenile roach were identical, therefore, the results will be reported together as the simulation outputs were also identical. These life stages have a preference for velocities up to 0.5 m s^{-1} and depths greater than 1 m (Armitage and Ladle, 1991). Prior to restoration, the results suggest that physical habitats suitable for adult and juvenile roach were highly abundant throughout the reach in deeper areas (Fig. 6.13). As the channel was uniformly deep and the river was slow flowing, the results suggest that 87 % of this physical habitat was coherent and classified as highly suitable (Table 6.5).

The simulations indicate that the physical habitat suitable for spawning roach (velocities between 0.4 – 0.8 m s^{-1} and depths of 0.25 – 3 m (Armitage and Ladle, 1991)) was less abundant. This physical habitat appeared to be restricted to the core of maximum velocity within the reach where velocities were locally elevated (Chapter 5). However, despite being low in abundance, 96 % of this spawning physical habitat was classified as highly suitable. The results also indicate that the provision of physical habitat suitable for fry (preferable depths between 0.1 – 0.3m and velocities up to 0.2 m s^{-1}) was of a high quality but very limited and fragmented prior to restoration.

Post-restoration (low flows)

Immediately following restoration, the results suggested that the area of suitable physical habitat for adult and juvenile roach decreased within the reach (Table 6.5). The decrease in physical habitat for these life stages was significant over the riffle feature as water depths were reduced and velocities increased outside of the preferable range for these life stages. This appeared to fragment the existing and extensive high-quality habitat (Fig.6.13). However, results indicate that the resultant smaller patches of high suitable habitat retained their high level of coherence observed prior to restoration (Table 6.6).

The simulations of the low flow 12-month post-construction survey data indicated that the extensive morphological adjustment had further modified the physical habitat configuration within the reach. As the riffle feature elongated it became a more extensive area of unsuitable physical habitat adult and juvenile roach (Fig. 6.13). Overall, the results suggest that the abundance of these physical habitats for these life stages increased within the reach. However, the quality of this physical habitat was likely to have been less preferable for these species following construction and readjustment of the riffle feature (Table 6.6).

The results suggest that the construction of the riffle feature locally increased physical habitat suitable for spawning roach by 550 % by elevating velocities up to 0.8 m s^{-1}, and 77% of this habitat was classified as highly suitable and coherent (Fig. 6.15 and Table 6.6). However, the suitability of the physical habitat over the riffle feature decreased over the crest as the peak velocities were less preferable for spawning roach (Armitage and Ladle, 1991). Following morphological adjustments within the reach during the 2013/14 flood events, the total suitable physical habitat for spawning increased by 47%. This physical habitat was of a higher suitability (96 %) and was more coherent as a greater area of velocities within the preferable range for spawning roach were experienced over the riffle feature (Fig. 6.15 and Table 6.6).

An increase in physical habitat suitable for fry was suggested by the simulation immediately following restoration, notably over the crest of the riffle feature as depths were decreased (Fig. 6.14). Nonetheless, the results indicated that this physical habitat was still in low abundance following restoration and it was also of a low suitability. Following the 2013/14 flood events, the riffle crest (although re-sited) may have still presented some suitable habitat for fry due to the shallow depths (Fig. 6.14). However, despite the results indicating the reach at this time provided the most abundant physical habitat for fry habitat throughout the monitoring period, it was still in low abundance and fragmented (Table 6.6).

Post-restoration (moderate and high flows)

During the moderate flow captured 3-months post-construction, the results suggest that both the adult and juvenile roach habitat was more abundant (~ 60% more) than at low flows immediately post-construction. Despite this increase in physical habitat, the overall suitability of the physical habitat decreased with a ~ 30% proportional decrease in highly suitable habitat. The simulations indicate that the highly suitable habitat was marginalised from the centre of the channel (Fig. 6.13). Despite a preferential increase in depth in this area, velocities increased above the preferential 0.5 m s^{-1} (Chapter 5.3). Whilst the abundance and suitability of these spaces may have decreased, the spatial metrics suggest that the spaces may have been more coherent (Table 6.5).

Following the 2013/14 flood events, the results suggest very similar patterns in physical habitat suitability for adult and juvenile roach habitat at moderate flows. These physical habitats were simulated in a similar abundance than observed prior to the floods at moderate flows. However, 9% more of this physical habitat was classified as high quality (Fig 6.15 & Table 6.5), possibly as a result of a preferential increase in depth and slight decrease in velocity which improved suitability over the upstream pool (Chapter 5.3). In comparison to the post-flood low flow, the physical habitat at both the moderate and high flows was overall of a lower quality, thus a similar trend to that observed prior to the floods (Table 6.5). Interestingly, the physical habitat suitability simulated over the pools within the reach at high flows decreased as velocities further increased with discharge.

Following the construction of the riffle feature (pre-flood), the results suggest that physical habitat suitable for spawning habitat increased at moderate flows (on the pre-flood low flow) by over 150%. Furthermore, 71% of this total physical habitat was classified as highly suitable and observed to be highly coherent. It is probable that this improvement in habitat suitability corresponds to an increase in velocity throughout the reach to within the preferential range for spawning roach (i.e. 0.4 – 0.8 m s^{-1}). However, over the riffle feature the velocities increased above this range (probably due to a decrease in depth) which may have contributed to an estimated decrease in suitability over this area.

The suitable physical habitat for spawning roach was less abundant following the 2013/14 floods, particularly in the pools where the average velocity decreased (Chapter 5.3). Additionally, the results indicate that the physical habitat was generally less suitable and more fragmented than the previous moderate flow. The spatial configuration of physical habitat was very similar to that observed prior to the floods with the riffle feature potentially affording a less suitable environment. A similar pattern was observed at higher flows, however, a slight decrease in velocity over the pools may have contributed to an increase in physical habitat suitability for spawning roach in these areas. The results suggested that there was no appreciable change in physical habitat provision for fry at moderate or high flows throughout the monitoring period. As estimated at low flows this physical habitat was in a low abundance and fragmented along the margins, the small amount of suitable physical habitat observed over the riffle crests was lost as discharge increased (Table 6.5 and Fig. 6.14).

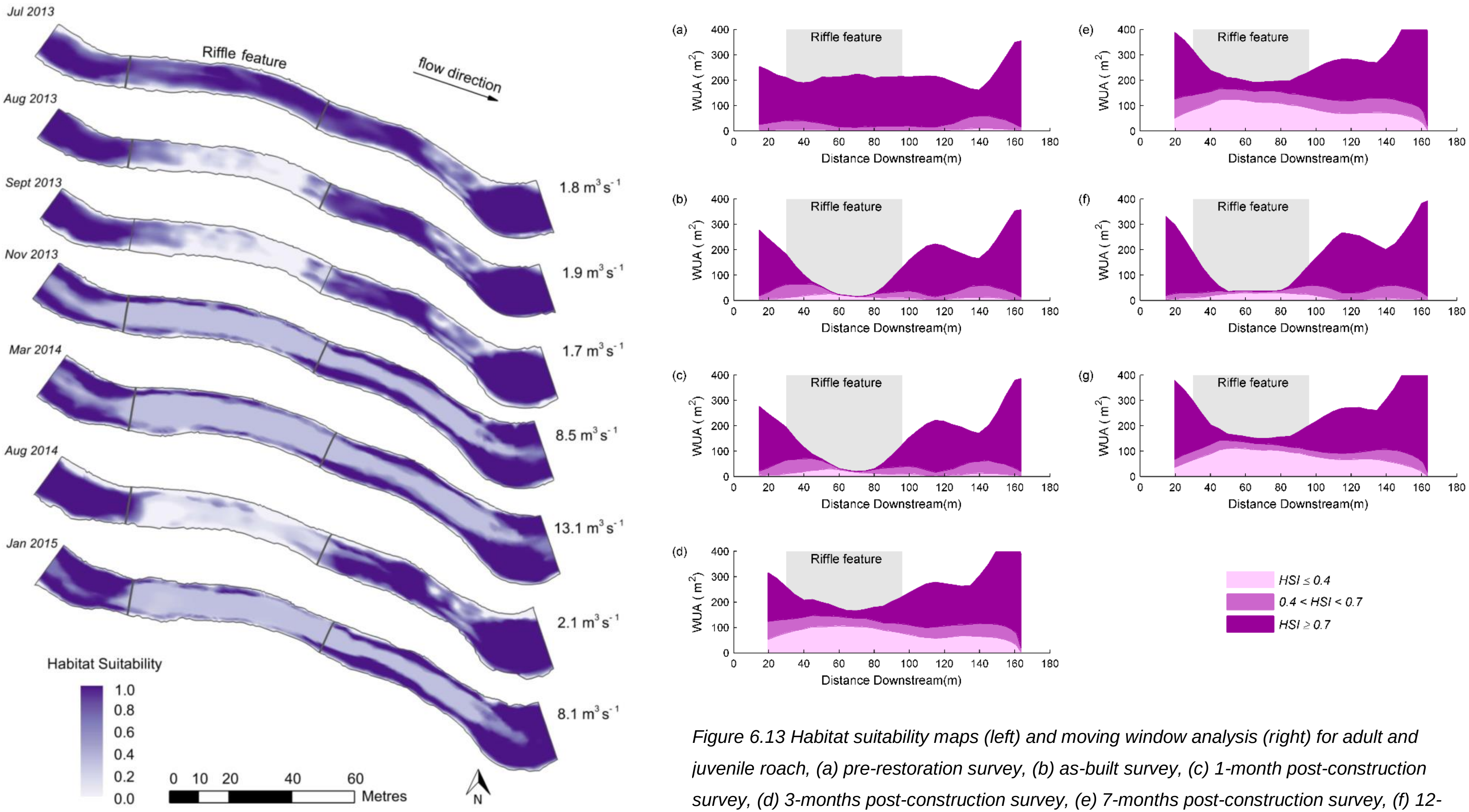
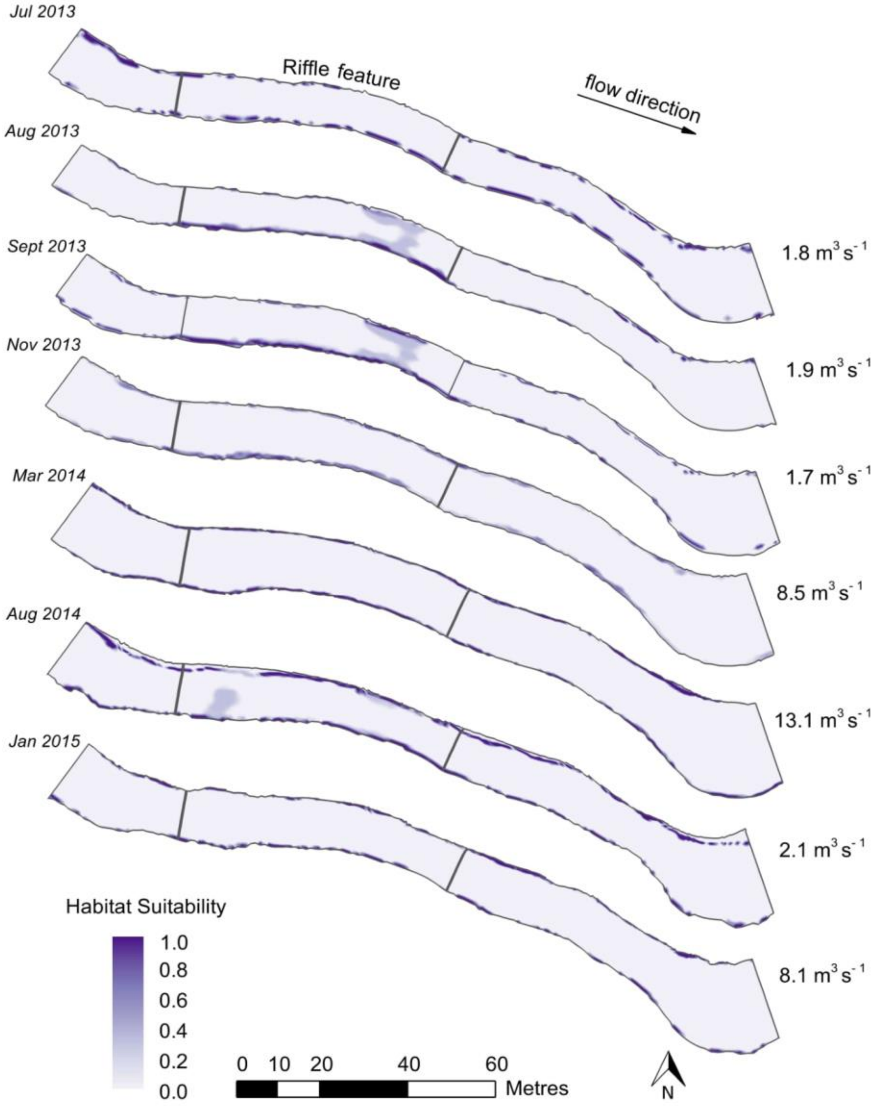


Figure 6.13 Habitat suitability maps (left) and moving window analysis (right) for adult and juvenile roach, (a) pre-restoration survey, (b) as-built survey, (c) 1-month post-construction survey, (d) 3-months post-construction survey, (e) 7-months post-construction survey, (f) 12-months post-construction survey and (g) 18-months post-restoration survey.



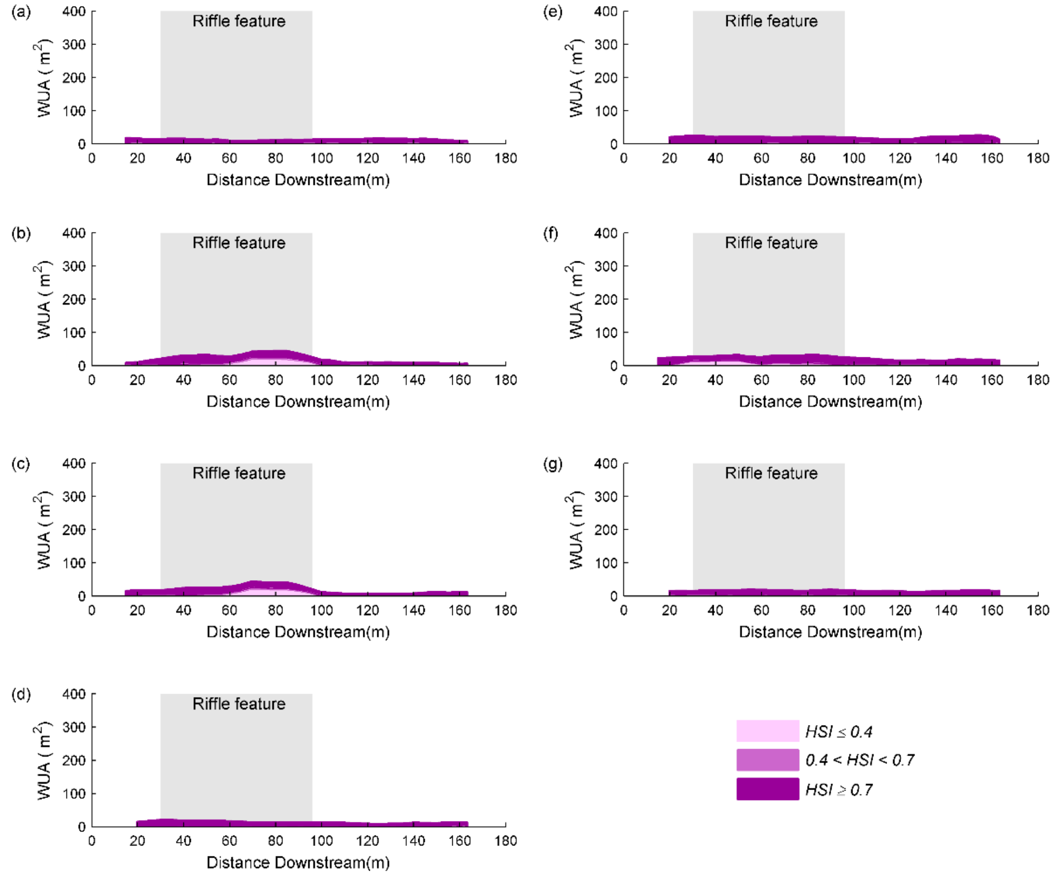


Figure 6.14 Habitat suitability maps (left) and moving window analysis (right) for fry roach, (a) pre-restoration survey, (b) as-built survey, (c) 1-month post-construction survey, (d) 3-months post-construction survey, (e) 7-months post-construction survey, (f) 12-months post-construction survey and (g) 18-months post-restoration survey.

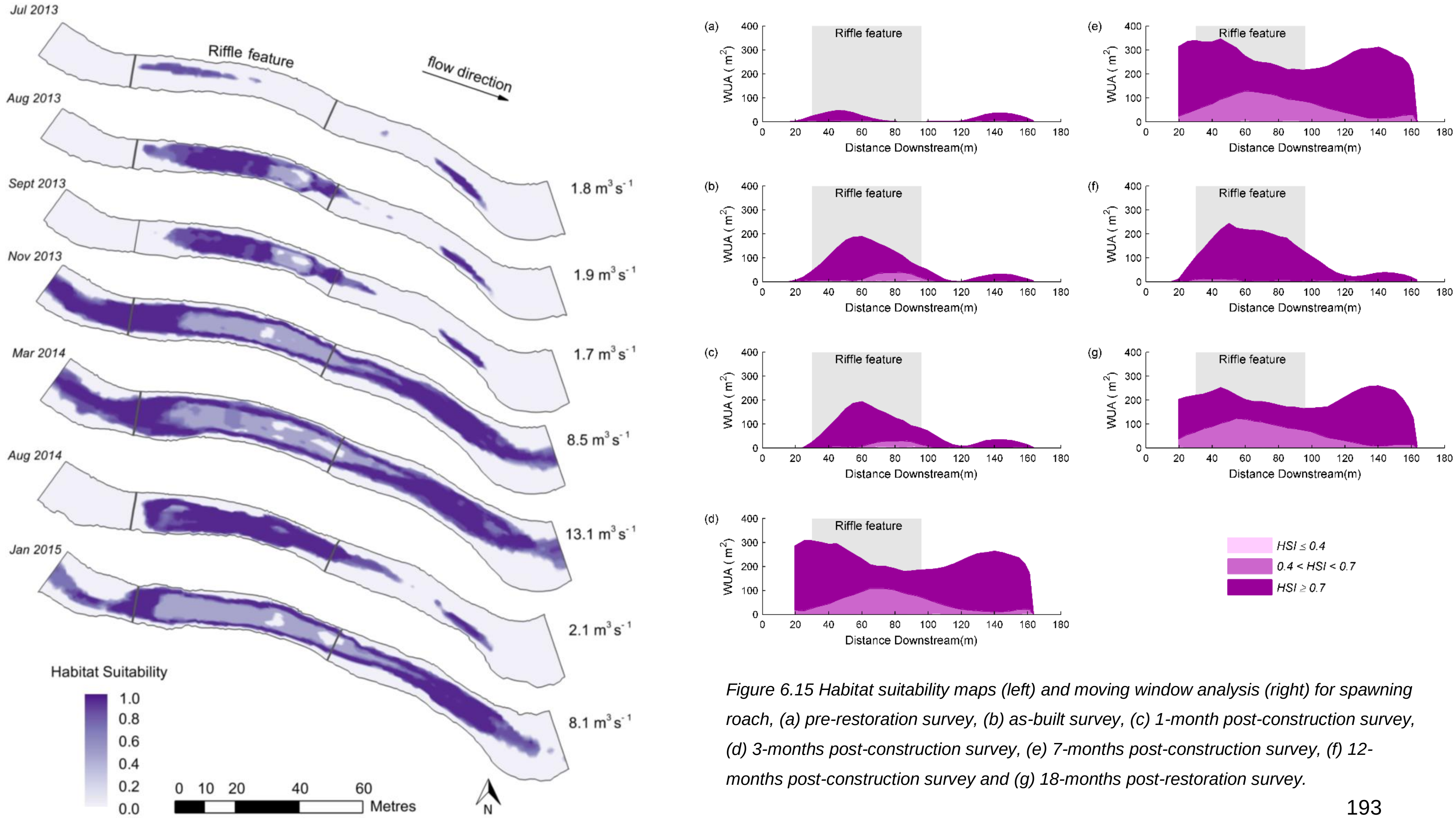


Figure 6.15 Habitat suitability maps (left) and moving window analysis (right) for spawning roach, (a) pre-restoration survey, (b) as-built survey, (c) 1-month post-construction survey, (d) 3-months post-construction survey, (e) 7-months post-construction survey, (f) 12-months post-construction survey and (g) 18-months post-restoration survey.

Table 6.5 Spatial metrics of habitat suitability for roach.

		Habitat Suitability											
		Low				Moderate				High			
	Total Area (m^2)	% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension
Adult and Juvenile Roach													
Baseline	1400	2	4	1.83	1.54	10	9	1.11	1.49	87	2	0.32	1.29
As-built	1286	18	6	0.92	1.46	16	7	0.98	1.47	66	5	0.33	1.26
1 Month	1371	19	8	0.77	1.41	16	13	1.04	1.50	65	5	0.33	1.27
3 Months	2036	47	2	0.42	1.35	19	24	1.60	1.69	34	6	0.83	1.52
7 Months	2363	48	3	0.32	1.29	18	14	1.41	1.66	33	6	0.75	1.50
12 Months	1589	25	7	0.64	1.39	12	11	1.33	1.58	63	3	0.29	1.24
18 Months	2047	47	1	0.36	1.30	10	21	1.77	1.69	43	6	0.53	1.41

Table 6.5 continued

		Habitat Suitability											
		Low				Moderate				High			
	Total Area (m^2)	% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension
Fry Roach													
Baseline	5	-	-	-	-	-	-	-	-	100	3	5.44	2.38
As-built	68	80	2	1.27	1.42	2	1	6.26	3.71	18	6	4.71	2.13
1 Month	61	95	2	1.13	1.38	-	-	-	-	5	2	8.23	3.41
3 Months	4	-	-	-	-	-	-	-	-	100	7	9.73	3.28
7 Months	14	-	-	-	-	-	-	-	-	100	20	11.02	2.78
12 Months	72	71	4	1.34	1.44	-	-	-	-	29	15	7.62	2.43
18 Months	5	-	-	-	-	-	-	-	-	100	5	9.99	3.20
Spawning Roach													
Baseline	56	-	-	-	-	4	3	6.12	3.07	96	2	1.69	1.57
As-built	363	4	2	1.54	1.28	19	3	0.97	1.33	77	3	0.66	1.36
1 Month	371	4	2	1.46	1.26	16	4	1.51	1.52	80	3	0.65	1.36
3 Months	1122	-	-	-	-	29	5	0.63	1.36	71	2	0.61	1.44
7 Months	1415	0.2	2	5.36	2.73	35	11	0.65	1.41	65	1	0.59	1.44
12 Months	534	-	-	-	-	4	5	2.25	1.63	96	3	0.45	1.30
18 Months	1029	-	-	-	-	44	6	0.61	1.39	56	6	0.84	1.51

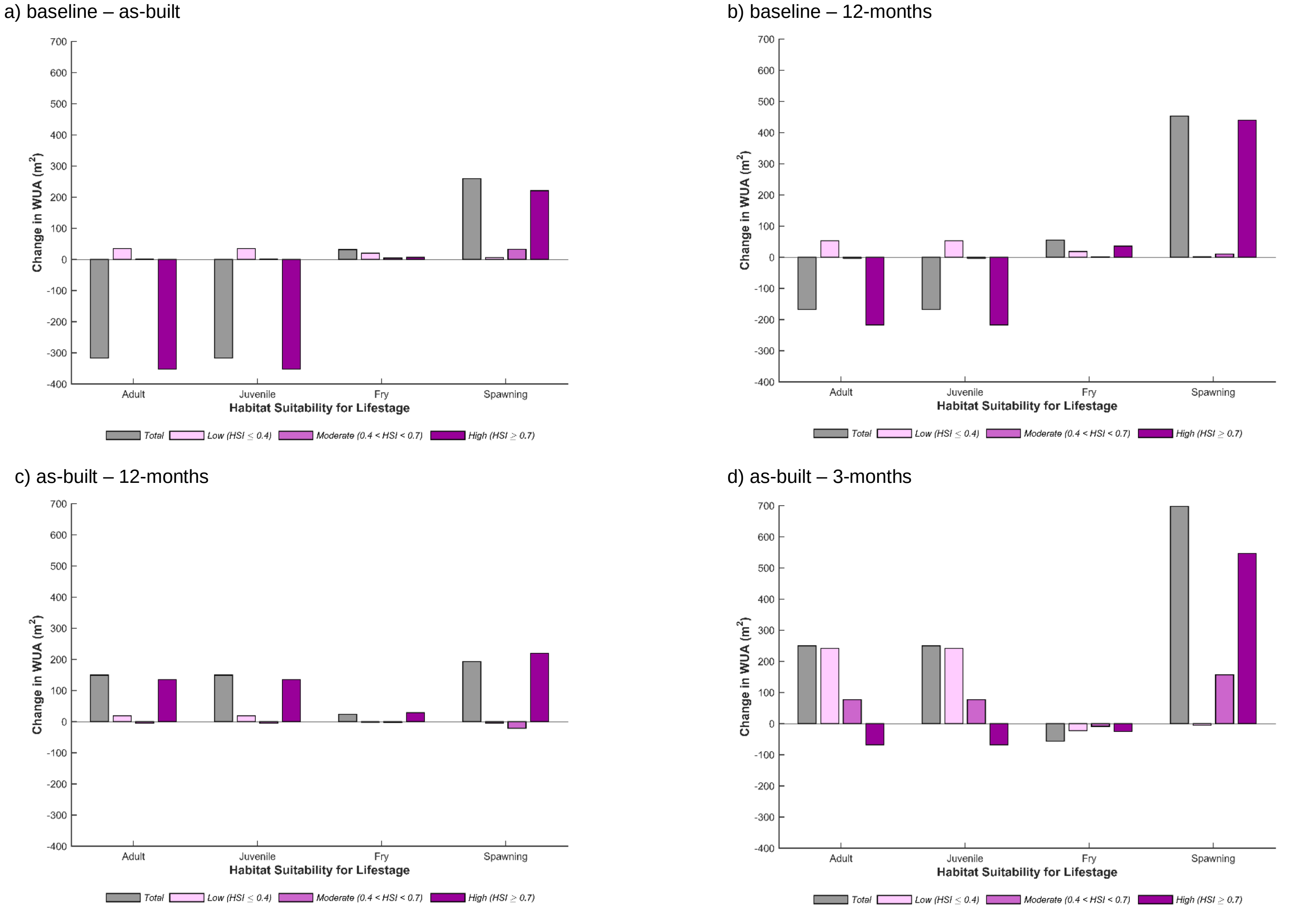
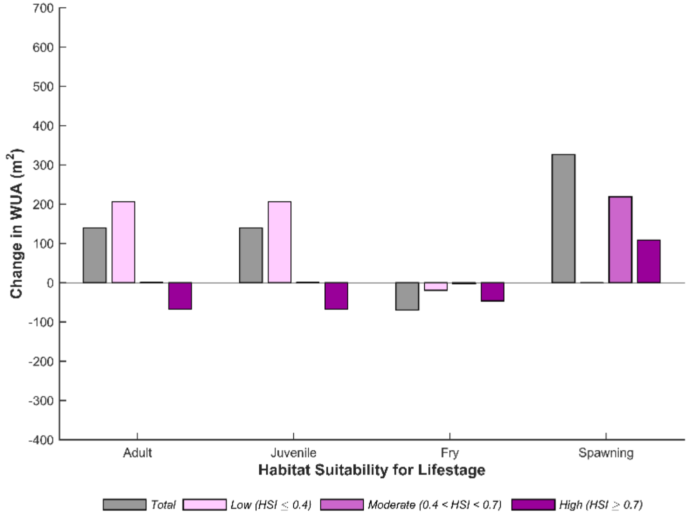
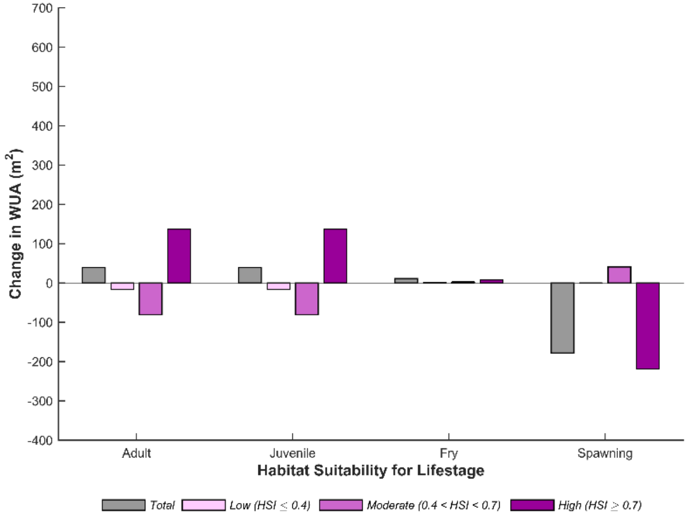


Figure 6.16 Change in WUA for roach between surveys a) baseline and as-built, b) baseline and 12-months, c) as-built and 12-months d) as-built and 3-months e) 12 months to 18-months, f) 3-months to 18-months g) 7-months to 18-months.

e) 12-motnhs – 18-months



f) 3-motnhs – 18-months



g) 7-motnhs – 18-months

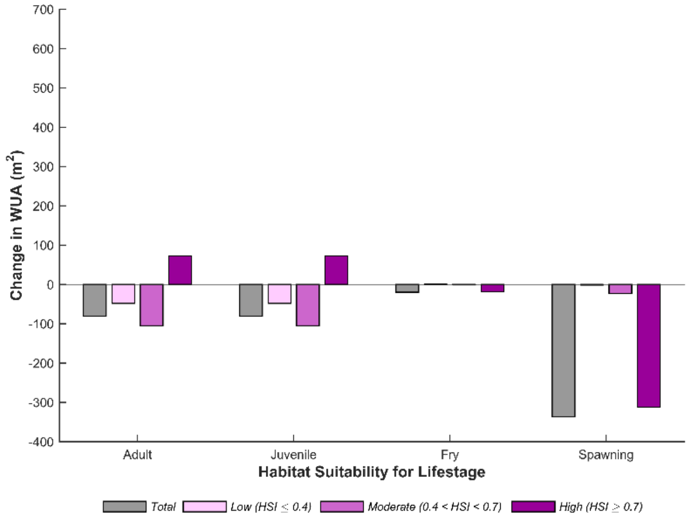


Fig. 6.16 (continued) Change in WUA for roach between surveys a) baseline and as-built, b) baseline and 12-months, c) as-built and 12-months d) as-built and 3-months e) 12 months to 18-months, f) 3-months to 18-months g) 7-months to 18-months.

6.4 Interpretation of physical habitat performance

This section reviews the results outlined in the previous two sections, in the context of baseline catchment data, to interpret the physical habitat performance of the RHES within the 18-month post-construction period. Initially, this section discusses the baseline monitoring results to interpret the physical habitat performance of the reach prior to restoration. These findings will provide a benchmark for discussing the post-construction performance of the scheme. The marked range of flows within the first 6 months following construction resulted in significant geomorphological changes that altered the configuration of physical habitat within the reach (Chapter 5). Therefore, these serendipitous events have presented an excellent opportunity to review the feature's resilience to extreme events.

It is not completely surprising that the interpretation of the scheme's physical habitat performance could also be influenced by the spatial scale over which the analyses are performed as this was observed within the geomorphological performance results (Chapter 5). However, this chapter does also highlight the importance of accounting for the spatial configuration of habitats when providing an assessment of performance. This is a characteristic which has not been considered in many physical habitat assessments possibly due to a lack of spatially continuous data (Chapter 2). This is a key finding that has significant implications for the design of future river restoration monitoring schemes, which will be illustrated in this chapter and explored more thoroughly in Chapter 7.

This section, when discussing the performance of the scheme, emphasises the effect of spatial configuration on the interpretation of performance and hence recommendations for adaptive management. The range of physical habitat assessment methods used also afford different interpretations on the schemes performance. Therefore, this serves to highlight the importance of using a suite of analyses when assessing the physical habitat performance of river restoration schemes. This section also reiterates the importance of collecting and using baseline data prior to restoration to inform achievable objectives and sustainable river restoration designs.

6.4.1 Pre-restoration performance

The results indicate that prior to restoration the study site, which was representative of the lower catchment, afforded limited PHH when assessed at the reach scale. At the sub-reach scale, the area in which the riffle feature was later constructed exhibited less PHH than the wider reach. It is possible that deposition in this sub-reach, potentially promoted by localised widening, contributed to the lower PHH in this area. This low PHH in the sub-reach was not mirrored by the MWA of HMID which suggested that some of the more heterogeneous spaces within the channel were afforded in this area. This was probably a reflection of micro-scale topography within the HMID assessment that uses the coefficient of variation. This example highlights the importance of considering scale when undertaking PHH assessments and the need for an improved understanding of the impact of PHH on the ecological status of rivers at different spatial scales.

Limited PHH has been linked to low recruitment of fish as the variety of physical habitats needed to support the full life cycle and food web are not necessarily available (Aarts et al., 2004; Miller et al., 2010). This offers a partial explanation for the absence of brown trout and low numbers of dace and roach recorded in Environment Agency fish surveys prior to restoration (EA, 2013). More specifically, the physical habitat simulations suggest that the reach afforded limited physical habitat suitable for brown trout in the spawning and fry life stages because the river was too deep. Shallow refuge habitat is critical for this life stage as they are poor swimmers in the initial stages of life (Elliot, 1987). The substrate was also assumed to be largely comprised of coarse sand (Chapter 3), which may have limited the survival of eggs where the hydraulic conditions were suitable for spawning (Armitage and Ladle, 1991). Salmonid eggs are laid in gravel to provide a sufficient supply of oxygen to facilitate growth (Greig et al., 2007).

The provision of physical habitats suitable for adults was also limited within the reach, as the velocities within the reach were too low. The provision of a high-quality adult habitat is critical to the success of the lifecycle as it can improve the fitness of older individuals, such as maternal fish. High maternal fitness has been linked to fecundity, egg size and habitat provisioning for the young which, in turn, affects the subsequent retention of the younger salmonid population (Einum and Flemming, 2000; Heath et al., 2003; Régnier et al., 2013). If brown trout managed to survive through the initial life stages, either within the reach or elsewhere within the catchment, then reach would have afforded abundant, highly suitable physical habitat for juvenile brown trout at low flows.

The physical habitat simulations of dace and roach also suggested that the physical habitat provision for the fry and spawning life stages were also limited. These two species differ from brown trout (a salmonid species) as they are cyprinids and are typically less migratory. These species also require marginal or refuge physical habitat in abundance as they are highly susceptible to displacement particularly during higher flows, additionally this physical habitat is also required in abundance as the competition for food is high, (Mills, 1982; Mann and Bass, 1997; Nunn et al., 2003; Nunn et al., 2007; Nunn et al., 2010).

The results for both dace and roach indicate that the physical habitat provision for adult and juvenile life stages in the reach was of a very high quality at low flows prior to restoration. This was more marked for roach as they prefer deeper and slower flowing environments which was characteristic of the reach prior to restoration. The presence of these species as recorded by the EA in 2013 prior to restoration does not necessarily indicate a high habitat quality in the lower catchment. This is because dace and roach are generalists that are able to colonise low quality environments (Beardsley and Britton, 2012a; 2012b; Murray, 2014). However, population structure and individual characteristics (e.g. growth rate and age) can provide an indication of general habitat quality. A lack of appropriate physical habitat and interspecific competition for food has been linked to slow growth, but prolonged life spans of these species (Weatherley, 1987; Nunn et al., 2007; Nunn et al., 2012). Additionally, substrate complexity (a known issue on the River Rother) has been associated with altered feeding patterns (Murray et al., 2016) which may potentially affect growth rate and the success of the population.

Slowed growth has been linked to a reduced fecundity and reduced egg size in this species, which has then been linked to subsequent egg mortality (Mann and Mills, 1985; Beardsley and Britton, 2012a). Thus, whilst these are generalist species, the size and quality of their populations are altered by physical habitat quality. Furthermore, a lack of appropriate refuge habitat for young cyprinids in periods of high flows has been linked to slow growth of the individual, through decreased water temperatures and increased energy expenditure required to survive in poor quality environments (Nunn et al., 2003). Growth data was not available for the year’s preceding restoration, so the state of the community prior to restoration cannot be discussed in this section. However, growth data was available for 2015 and thus will be discussed later in the chapter in relation to the post-flood design.

The ADCP data has afforded detailed interpretations of in-channel geomorphological and physical habitat performance within the reach. However, this data can be used in conjunction with other high-resolution datasets and qualitative observations of wider catchment processes to inform broader interpretations of physical habitat performance. Young fish are vulnerable to lateral displacement onto the floodplain during flooding and becoming stranded as flows recede if there is a lack of floodplain connectivity (Jensen and Johnsen, 1999; Bolland et al., 2015). The banks of the reach are sufficiently higher than the adjacent floodplain (a potential relic feature of the Rother Navigation), therefore, the potential for re-entry by individuals during high flow events may have been limited. This would suggest that during out-of-bank flow events, lateral dis-connectivity in the reach may have contributed to a reduction in the population of fish species (Aarts et al., 2004). Restoration of floodplain connectivity is critical to increasing the resilience of the river ecosystem, particularly for specialist species, as climate change predictions have forecasted an increase in flood frequency and magnitude in the future (Watts et al., 2015).

Additionally, the presence of two significant migratory barriers on the River Rother downstream of the riffle feature may have also contributed to the low recruitment of brown trout. The reach provided a high quality physical habitat for juvenile brown trout prior to restoration, but these barriers are also bad for juveniles because they can be displaced in a high flow event and find themselves unable to migrate back to this upstream habitat. The importance of barrier easement for juveniles as well as adults has been recently highlighted to aid post-flood recovery (Tummers et al., 2016; Forty et al., 2016). Furthermore, higher migration costs may result in a lower fecundity in adults (Bohlin et al., 2001), and may further reduce the success of spawning when combined with the limited physical habitat availability afforded within the reach. Although one of these two sites, Hardham Weir, had undergone a fish easement programme prior to the restoration of the reach, Fittleworth Mill still presented a significant barrier to migratory fish (ARRT, 2017).

Consequently, prior to restoration, the evidence suggests that the River Rother was a poor physical habitat for a range of fish species. The pre-restoration survey and consideration of wider factors have highlighted that prior to restoration there were potentially multiple of factors in addition to a lack of spawning habitat which may have impacted on the life cycle of brown trout. The restoration of in-channel physical habitat for fry, spawning and adults would have been essential for improving the recruitment of brown trout. Whilst in-channel physical habitat was likely to be a major contributor to the low abundance of brown trout within the catchment, other factors such as floodplain connectivity were also likely to have been contributing factors. The reach did afford an excellent habitat for some life stages of the cyprinid species and intervention may have resulted in a degradation of this physical habitat. However, shallow marginal habitats suitable for the younger life stages were lacking and likely to have been affecting the recruitment of these species. Therefore, this study supports previous studies (e.g. Downs and Kondolf, 2002; Downs et al., 2011) which highlight the value of baseline assessments to guide the development of project objectives and pre-restoration design.

6.4.2 Post-restoration performance

The post-restoration performance of the RHES is discussed in relation to the performance before and after the geomorphologically significant 2013/14 flood events. The relative improvements to physical habitat within the reach are summarised in Table 6.6.

Physical habitat performance prior to the 2013/14 floods

Immediately following the construction of the riffle feature, the HMID score increased which suggests that physical habitat heterogeneity improved at low flows. Additionally, the clustering analyses indicated a greater diversity of physical habitats were afforded by the reach than prior to restoration. This would suggest that the physical habitat performance improved at low flows as a more diverse habitat condition typically supports a more diverse community (Gostner et al., 2013). Although a primary driver of restoration is to improve fish habitat, habitat heterogeneity is particularly important in for supporting different stages of the life cycle and the food web. For example, Miller et al. (2010) noted that improvements to habitat heterogeneity had a significant positive impact on the richness of invertebrate species. Therefore, the reach had the potential to support a more diverse community at low flows but also presented opportunities to improve prey for consumers, such as fish, within the food web.

The HMID results suggest that the reach afforded less PHH at moderate flows prior to the 2013/14 flood events. On the other hand, the hydraulic clustering analysis suggested that the PHH improved at moderate flows as the diversity between physical habitat spaces increased. Therefore, the results do not provide a clear interpretation of physical habitat performance of the reach at moderate flows prior to the 2013/14 flood events. It does, however, serve to highlight that different physical habitat assessment methods may afford

*Table 6.6 Summary of results outlined in Sections 6.2 and 6.3, the symbols denote physical habitat performance as either a → slight improvement, ➔ significant improvement, = no discernible change, → slight deterioration or ➔ significant deterioration. The asterisks denote the comparison between surveys as either * comparable low flows ** low to moderate flow comparison, *** comparable moderate flows **** moderate to high flow comparison.*

	Habitat Heterogeneity Assessment		**Physical Habitat Simulation**											
			Brown Trout				Roach				Dace			
	HMID	HCA	ADULT	JUVENILE	FRY	SPAWNING	ADULT	JUVENILE	FRY	SPAWNING	ADULT	JUVENILE	FRY	SPAWNING
Baseline to as-built *	↑	↑	↑	↑	↑	⬆	⬇	⬇	↑	⬆	=	⬇	↑	⬆
Baseline to 12 Months *	⬆	⬆	⬆	↓	↑	⬆	⬇	⬇	↑	⬆	=	⬇	↑	⬆
As-built to 12 Months *	↑	↑	↑	↓	=	⬆	=	=	↓	↑	=	⬇	↑	⬆
As-built to 3 Months **	↓	↑	⬆	⬇	↓	↓	=	=	↓	⬆	↓	⬇	↓	⬇
12 Months to 18 Months **	⬇	↑	⬆	↓	↓	↓	=	=	↓	↑	↓	↓	↓	⬇
3 Months to 18 Months ***	↑	↑	↑	=	=	↑	↑	↑	=	↓	↓	=	↑	↑
18 Months to 7 Months ****	↓	↑	⬇	↓	=	⬇	↓	↓	=	⬆	↓	=	↑	↓

different interpretations of performance. This finding has significant implications for river restoration monitoring practice as it would suggest that multiple assessment methods are needed to interpret physical habitat performance.

The conflicting results highlight the value of using multiple approaches to appraise river restoration success. For example, the results of the physical habitat simulations suggest that although it was not a direct objective of the scheme, the physical habitat provision for all life stages of brown trout improved in abundance at low flows through the construction of the riffle feature. However, the most marked improvement was seen in the provision of physical habitat suitable for spawning trout. The abundance of brown trout spawning habitat, which was a direct objective of the scheme, was increased by nearly 250% at low flows potentially through the decrease in water depth over the riffle feature.

The abundance of this brown trout physical habitat was maintained at moderate flows but the suitability decreased. This would suggest a decrease in physical habitat provision as discharge increased, which reflects the observations of the geomorphological performance assessment reported in Chapter 5. This suggests that the riffle feature became less influential as discharge increased, as reflected in the HMID assessment. However, given that this spawning habitat was observed over the riffle feature it is conceivable that this physical habitat may not have been provisioned prior to intervention within the reach. Therefore, it is possible that the physical habitat performance for the target species was also improved at both moderate and high flows following restoration. Nonetheless, the performance likely decreased as discharge increased.

Some morphological changes occurred within the reach between the as-built and 3-month post-construction surveys which may have had implications for physical habitat provision. These changes include a small but significant amount of deposition over the riffle feature within this period, and this deposition (probably comprised of sand) may have been promoted by the riffle feature (Chapter 5). If this deposited material was largely comprised of fine material, it could have implications for the quality of physical habitat provision for spawning brown trout. As discussed in the previous section, salmonid eggs are laid in gravel to provide a sufficient supply of oxygen to facilitate growth (Greig et al., 2007).

The deposition of fine sediment within gravels limits the supply of oxygen to the eggs, and depending on the degree of sedimentation of the gravels, can result in egg mortality (Sear et al., 2016). In addition, whilst some alevin (newly hatched salmonid) can emerge from gravels through up to an 8cm deposit of sand (Crisp, 1993), in these poor habitat conditions alevins emerge sooner after spawning and are subsequently of a lower fitness

(Rubin, 1998; Sear et al., 2016). The deposition over the riffle feature was typically observed within the 1-3 month period to be less than 10 cm. Therefore, in a worst-case-scenario of blanket sand deposition over the feature, the riffle feature may still have been a viable egg incubation habitat, but would probably have resulted in individuals with a lower fitness emerging from the reach.

The riffle feature was not directly designed to afford physical habitat for fry, however, the literature suggests that fish within their early life stages require suitable coherent habitats in close proximity to each other (Aarts et al., 2004). Nearly a 60% increase in shallow slow flowing environments suitable for fry were observed at a low flow following restoration. However, although the fry habitat was more abundant it was still relatively scarce and was found to be highly fragmented. Furthermore, the results suggest that the physical habitat provision for the younger life stages decreased during moderate flows. Even in a best-case scenario whereby the deposition of material over the suitable spawning areas within the reach comprised of gravel, the limited physical habitat provision within the reach suitable for fry potentially reduces the chances of survival for both alevin and fry.

Focused restoration of physical habitat for fry has been found successful in improving recruitment in later life stages (Palm et al., 2010), as this stage of the life cycle is the most competitive (Elliot, 1986; Gauthey et al, 2015). Immediately following emergence, fry stay within the local vicinity of their spawning site as they are poor swimmers in this life stage (Elliot, 1987). Therefore, there is a strong case for restoring fry and spawning habitats in close proximity to each other to strengthen the probability of recruitment to the juvenile life stage (Armstrong et al., 2003). The maximum distance young brown trout are likely to disperse to seek appropriate habitat is thought to be a few hundred meters from the spawning site (Armstrong and Nislow, 2006). In the context of the current study, a large fish refuge was constructed approximately 300m upstream of the study reach. This is likely to be at the furthest extent of their migratory capabilities and habitat creation for this life stage should be considered nearer to the riffle feature in order to maximise the benefits it could afford.

Young brown trout are at a high risk of displacement from even low water velocities, and therefore typically disperse towards habitats in the downstream direction (Ottaway and Clarke, 1981; Ottaway and Forrest, 1983; Heggenes and Traaen, 1988; Daufrense et al, 2005; Andersson, 2016). The velocity patterns within the reach during both low and moderate flow exceeded the preferable conditions of 0.3 m s^{-1} (Armitage and Ladle, 1991). This suggests that young trout would have been highly likely displaced downstream.

Therefore, if the refuge habitat, placed upstream of the study reach, was constructed downstream of the riffle feature it would likely have been more successful in recruiting from it and improved the performance of the feature. That said, this upstream fish refuge would have still supported spawning habitats found even further upstream such as those implemented as part of the RHES on Burton Mill Stream. Immediately following construction, no obvious refuge habitat was observed downstream of the feature, and thus the retention of these young brown trout would have likely been limited without further intervention.

The results also suggest that the construction of the riffle feature improved the abundance and quality of spawning habitat for both cyprinid species at low flows within the reach. This physical habitat provision was not maintained for dace at moderate flows as the increased water depth drowned out suitable habitat. This would have likely had a high impact on the recruitment of dace as the species has been observed to spawn in early spring when flows are likely to be more fluctuant (Mann, 1974). The spawning physical habitat for roach significantly improved at moderate flows following restoration. This may have had less of an impact for roach as they spawn in late spring when flows are typically lower (Mann, 1973). However, the spatial configuration of this physical habitat suggests that this was unlikely to have been a result of intervention within the reach. It is possible that the riffle feature locally increased velocities to a range which was less suitable for this life stage.

The potential for deposition of fine sediment over the gravel of the riffle feature following restoration could have reduced the habitats suitability for rearing. Egg mortality and hatching success for these species have also been negatively linked to siltation in rivers (Mills, 1980; 1981). The deposition within the as-built to 3-month time period (potentially due to the design of the riffle feature as discussed in Chapter 5) was observed outside of the spawning season. It is important to note that such events may occur at any time within the year, but the spawning seasons for these cyprinid species (Table 6.7) are likely to follow a period of higher flows where fine sediment has been mobilised. Again, the impact of this on dace may be more severe than roach as some of this sediment may be cleansed from the feature as low flows start to prevail later in spring when roach eggs are likely to have been laid (Mann, 1973). The physical habitat provision for the fry of both of these cyprinid species following restoration was poor despite a slight improvement at low flows. This mirrors the trend seen in the provision of brown trout physical habitat and would suggest that the consideration of creating shallow marginal habitats nearer to the riffle feature would also be beneficial for these cyprinid species.

Table 6.7 Seasonal patterns of habitat use inferred from the angling forums. Shaded text indicates the months in which the surveys where captured and used to inform the physical habitat overlap index.

	January	February	**March**	April	May	June	**July**	**August**	**September**	October	**November**	December
Brown Trout												
Adult	**X**	X	**X**	X	X	X	**X**	**X**	**X**	X	**X**	X
Spawning	**X**	X									**X**	X
Juvenile	**X**	X	**X**	X	X	X	**X**	**X**	**X**	X	**X**	X
Fry			**X**	X	X	X	**X**	**X**				
Roach												
Adult	**X**	X	**X**	X	X	X	**X**	**X**	**X**	X	**X**	X
Spawning				X	X							
Juvenile	**X**	X	**X**	X	X	X	**X**	**X**	**X**	X	**X**	X
Fry						X	**X**	**X**	**X**	X	**X**	
Dace												
Adult	**X**	X	**X**	X	X	X	**X**	**X**	**X**	X	**X**	X
Spawning			**X**	X	X							
Juvenile	**X**	X	**X**	X	X	X	**X**	**X**	**X**	X	**X**	X
Fry						X	**X**	**X**	**X**	X	**X**	

Following restoration, the water surface of the riffle feature was noticeably more disturbed which had the potential to improve the oxygenation of the water. This may diversify the food supply, although these species are able to survive in poor environments due to an ability to feed on organisms that thrive in anaerobic conditions (Beardsley and Britton, 2012a; 2012b). Whilst these anaerobic species present a poor food supply and may limit the growth rate of the predator, water quality improvements have also been linked to a decline in roach and dace due to a diminished food supply (Beardsley and Britton, 2012a; 2012b). Therefore, given the uncertainty in habitat heterogeneity for improving ecological performance (Miller et al., 2010; Palmer et al., 2010; Pedersen et al., 2014), continued ecological monitoring of the reach would be advisable to monitor the impact of physical habitat alteration on these two species which were both observed in the lower catchment prior to restoration.

Despite this loss of physical habitat for adult and juvenile roach through the restoration design of the reach, the construction of the riffle feature was unlikely to have negatively impacted on the populations of these two species. This is because the reach (prior to restoration) was morphologically representative of the lower catchment, physical habitat was likely to have been found in sufficient abundance outside of the reach. Nonetheless, future restoration in the area should carefully consider the impact on the wider ecological community which may be generalists as well as the specialist target species.

The physical habitat suitability models can be used to assess the changes in physical habitat overlap and the potential for competition. The overlap between habitats was evaluated for each survey dependant on the species and life stages (reported here) that were likely to have been using the reach at the time of data collection (see Table 6.7). A score of 0 indicates a low overlap between physical habitats and 1 indicates a maximum overlap between habitats (i.e. the physical habitat is suitable for all species that may seasonally be using the reach). This additional analysis (Fig. 6.17) indicates that the construction of the riffle feature increased the competition in the channel margins over the feature during summer low flows. Similarly, this assessment indicates that competition for shallow marginal habitat was also high between these species during moderate flows. Individually the physical habitat suitability models suggested that marginal habitat was scarce for each species. This highlights that as these species have similar habitat preferences there may have been high interspecific competition for physical habitat following restoration.

In summary, the design of the reach immediately post-construction of the riffle feature not only enhanced the spawning habitat of the target species of brown trout but also improved the spawning habitat of two cyprinid species observed within the reach prior to restoration. These improvements were observed at both low and moderate flows for brown trout and roach, yet only at low flows for dace. The composition of the material which was deposited over the riffle feature (as discussed in Chapter 5) is unknown but evidence from the literature suggests that a small amount of fine sediment can be tolerated during the incubation and emergent period of all the species lifecycles. However, the fitness of the individuals in the fry and later life stages may be affected as a result.

The reach afforded abundant physical habitat following restoration for the adult life stages. However, the current study suggests that for all species studied, there is a lack of coherent refuge habitat within and downstream of the reach to support intermediary life stages. Therefore, the full benefit of the riffle feature may not have been fully realised. Furthermore, the construction of the riffle feature directly modified the physical habitat provision and had the potential to negatively affect the cyprinid species, which were recorded within the catchment prior to restoration. This highlights the importance of considering and potentially compensating for other species when altering physical habitat. Ecological monitoring would have been required to validate the observations made from the monitoring of physical habitat, as the wider habitat is influenced by a wide range of abiotic and biotic interactions.

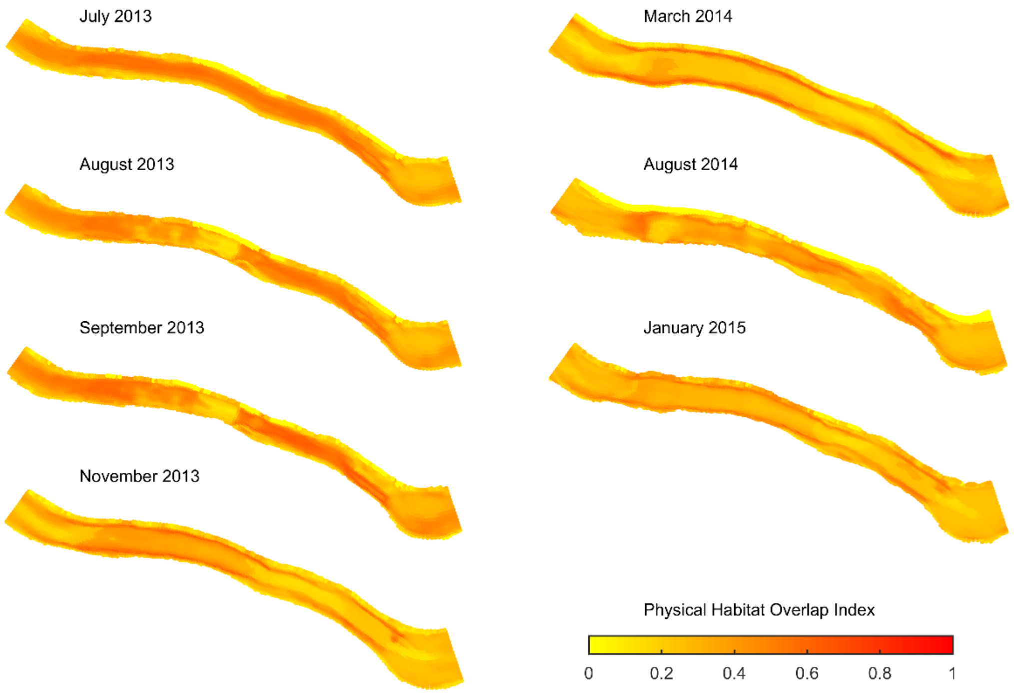


Figure 6.17 Overlap between physical habitats of the target species according to the time of year data was collected.

The geomorphological design of the riffle feature was largely resilient up to a bankfull flow, and the physical habitat abundance and quality of habitat for the spawning brown trout life stage improved as a result of this design. However, the potential access to this spawning site remained unchanged with respect to the pre-restoration condition. Furthermore, wider issues in affording refuge habitat and resilience during flows also remained unchanged. Further intervention would likely have been needed to reduce recruitment issues. Therefore, the construction of refuge habitats in close proximity to and downstream of future spawning habitat enhancement features, the consideration of further fish easement and floodplain reconnection would have been recommended in the absence of the changes seen following the 2013/14 flood events.

Physical habitat performance following the 2013/14 flood events

Chapter 5 discussed the significant geomorphological changes observed within the reach which resulted in the re-siting of the riffle crest and the deepening of two pools within the reach. The HMID scores suggested that the PHH afforded by the reach improved at both low and moderate flows (Table 6.6). However, the effect of the post-flood riffle feature in providing habitat heterogeneity still decreased as discharge increased. Conversely, the results of the hierarchical clustering suggest that a greater number of different hydraulic patches (that were also more coherent) emerged as discharge increased. A study of a reach on the River Arrow, Worcestershire found that moderate flow conditions provided the most advantageous configuration of hydraulic patches (Wallis et al., 2012). Similarly, variability between hydraulic patches in a study of the River Tern, Shropshire was maintained as discharge increased (Emery et al., 2003).

The findings of this study reported in this chapter indicate that the configuration of hydraulic patches was most advantageous during a high flow event. This supports the observations by Wallis et al. (2012) that hydraulic patches during low flow conditions can be more fragmented. The results of the RHES indicated that generally hydraulic patches observed in the channel margins (which are often associated with refuge habitat) became more fragmented as discharge increased. However, the hydraulic clustering analyses suggested that following the modifications to the reach during 2013/14 flood events more coherent marginal habitat was available during both low and moderate flows. This finding highlights the variable interpretations that these analytical methods could afford, the implications of such are discussed in Chapter 7.

The physical habitat simulations suggest that despite morphological adjustments, the physical habitat created for brown trout as part of the scheme was maintained if not improved following the re-working of the riffle feature. However, the simulations indicated that the structural modifications observed within the reach did not afford a greater provision of shallow marginal physical habitat for younger life stages. Some minor improvements to fry habitat were made in the channel margins at low flows which is good as fry typically emerge in summer when low flows prevail. However, moderate and high flow events may occur irrespective of season and thus younger brown trout may have been particularly vulnerable to summer flooding. Consequently, following the 2013/14 floods, it was unlikely that the reach supported the recruitment of brown trout to the later life stages.

Furthermore, the geomorphological monitoring results (Chapter 5) indicate some low levels of scour and fill during which could respectively correspond with the spawning and incubation phases of the brown trout life cycle. As discussed earlier in this section, the addition of fine material during the incubation/emergent stage (spring) is not necessarily completely deleterious (Crisp, 1993), however, it could impact on the fitness of individuals in later life (Sear et al., 2016). Unfortunately, monitoring of fine sediment storage within gravels following the 2013/14 flood events has suggested that the riffle feature had become infiltrated with fine sediment (Evans et al., 2017). Conversely, scour processes observed during spawning (late autumn/winter) phases could be advantageous through cleansing fine material from the spawning habitat. Although scour has also been observed to be detrimental to the life cycle as eggs can be removed from the incubation habitat if scour exceeds the burial depth (Tonina et al., 2008; Mckean and Tonina, 2013; Goode et al., 2013). Larger and fitter individuals in the population have been noted to bury their eggs deeper, thus it is critical to ensure the fitness of adults through provisioning adequate habitat conditions and removing barriers that could induce migratory costs. This is especially important as scour process are predicted to become more deleterious with climate change and eggs buried deeper may have a better chance of survival (Goode et al., 2013).

The EA 2015 fish monitoring survey found brown trout at all of the five sampling sites within the catchment (EA, 2015). These sampling sites include Coutlershaw and Fittleworth which are approximately 1 and 2km upstream and downstream of the study reach respectively (Chapter 3). These fish were found to be anadromous adults which had returned in multiple years to spawn, with one fish documented to be returning for its fourth run (EA, 2015). Therefore, the barriers in the catchment between the English Channel and the reach must be passable to at least some larger fish. It was noted, however, that all the fish caught were classified as adults and some were observed to be in a poor condition, which is of concern with regards to both the aforementioned scour and fill processes. This could be contributed to by one or more of the following factors:

- The limited provision of shallow refuge physical habitat for younger brown trout and other non-physical habitat parameters may have affected recruitment into the later life stages of the life cycle.
- Barriers in the wider catchment may have prevented the upstream migration of smaller individuals of the species,

- Barriers in the wider catchment may have introduced migratory costs on adults which negatively impacted on the survival and fitness of the young.

Following the 2013/14 floods, the habitat simulations for the cyprinid species conveyed mixed results. Both species saw improvements to spawning habitat at low flows in comparison to the constructed riffle feature. However, the shallow refuge physical habitat provision for fry was still found to be incoherent and in low abundance. The riffle feature, both before and after modification, fragmented the existing high-quality habitat for some of the life stages of the cyprinid species observed prior to restoration. This effect appeared to have intensified as velocities became more varied following the 2013/14 flood events. The 2015 fish monitoring survey still found dace and roach in the lower catchment following restoration, and these species were the most abundant in the lower catchment (EA, 2016). In addition, a local angling club reported significant roach and dace catches in the lower catchment during summer 2016 (Petworth and Bognor Angling Club, 2016). These reports would suggest that the cyprinid species have not been immediately negatively impacted by the restoration works in the local area.

In the 2015 fish survey, the population of dace was dominated by adults and the growth rates were below average, most notably for individuals less than three years old (EA, 2015). As discussed earlier in this section, slow growth rates could suggest that the lower catchment is affording a poor-quality habitat (Nunn et al., 2003; Nunn et al., 2007), and/or sedimentation has impacted on feeding patterns (Murrary et al., 2016). The cause of the slow growth rate warrants further scientific investigation to identify an appropriate course of restoration to improve the fitness of the cyprinid population within the wider River Rother catchment.

The habitat suitability models indicated that the riffle crest both as designed and as observed post-flood presented a lower quality hydraulic habitat than the rest of the riffle feature. These results support the findings of habitat simulations of pool-riffle designs which suggested that riffle crests of accentuated pool-riffle topographies offer low quality habitats (Pasternack and Brown, 2013). Accordingly, the analysis of physical habitat overlap (Fig. 6.17) suggests that competition for this area of the channel would have been quite low. In the wider reach, however, competition for marginal physical habitats increased with discharge (Fig. 6.17). The post-flood crest afforded a more gradual transition in elevations along the feature and the suitability models suggested that this newly formed topographic high was typically more suitable than the constructed crest. This may suggest that the species evaluated here may prefer a more gradually transitioning pool-riffle topography. However, further ecological monitoring would be required to validate this observation. This does not necessarily support the restoration design principle that a more heterogeneous physical habitat is more advantageous.

In summary, following the 2013/14 flood events, the reach further enhanced the spawning habitat of the target species (brown trout) and therefore was successful in achieving its objective after 18 months post-construction. The provision of physical habitat to support the recruitment of this species into later life stages was limited. In the creation of physical habitat for brown trout, the physical habitat provision for the cyprinid species evaluated here further deteriorated. This effect was not apparent immediately on the wider populations of these species which may suggest that there was suitable physical habitat elsewhere within the lower catchment. This study serves to highlight the importance of monitoring to identify any unintended adverse effects of restoration. Ecological monitoring, which is beyond the scope of this research, would have been beneficial to validate the observations made from physical habitat monitoring.

Overall, the riffle feature was fairly resilient to extreme events and met its objective of improving spawning habitat provision for brown trout. However, 18 months following the construction of the feature the access and the creation of complementary habitats to the improved spawning area remained unchanged with respect to the pre-restoration condition. Therefore, the recommendations which were made earlier in this section still stand based on the assessment of performance following to 2013/14 flood events.

6.5 Recommendations for the River Rother and similar environments

Since the 2013/14 flood events, further restorative measures have been implemented which may have positively benefited the ecological community, particularly through the provision of refuge habitats. Continued ecological monitoring will be required to explore the long-term performance of these measures in terms of supporting the recovery of the ecological community. These measures have included the construction of fencing and tree planting along the right bank of the riffle feature to limit livestock access to the channel. This has had a noticeable impact on the riparian community and reduced the potential for bank erosion at this location, as the trees mature they should have a noticeable impact on the provision of shade (Fig. 6.18).

Furthermore, as the riparian vegetation develops it may reduce velocity in the channel margins by increasing resistance (Malkinson and Wittenberg, 2007; Gurnell, 2014). This may promote the development of a more coherent area of slower flowing and shallower marginal physical habitat. This type of physical habitat was found in low abundance in the RHES monitoring campaign. This provision of this physical habitat may be particularly beneficial to improve the retention of fry and juvenile life stages. The replication of similar schemes along the River Rother and similar environments are recommended.

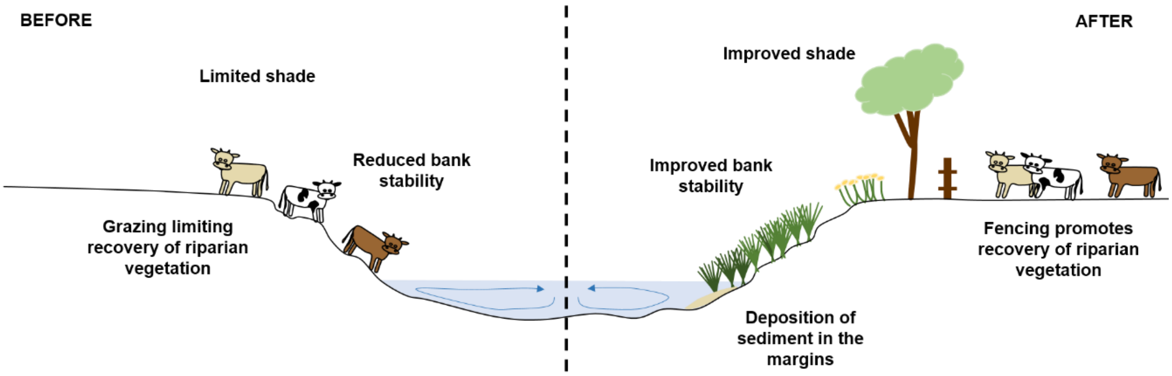


Figure 6.18 Potential impacts of unfenced grazing (before) and riparian restoration (after).

In addition, two fish refuges were constructed downstream of Fittleworth Mill in 2014 and were being utilised during a walkover survey in the summer of 2015 (Fig. 6.19) (Cox and Soar, 2017). These fish refuges could provide a valuable refuge habitat if young fish are displaced downstream of Fittleworth Mill in high velocities and are unable to migrate back upstream. However, given the in-channel sediment dynamics, it is likely these habitats will infill and require regular maintenance. Different approaches were used on each of these fish refuges; one was fenced off from livestock and the other was left open. The unfenced fish refuge was maintained but the fenced fish refuge exhibited strong evidence of infilling (Fig. 6.20). Therefore, by limiting livestock access to the river where refuge habitats are created, the longevity these habitats may be improved and widespread bank erosion may be controlled. A fish easement programme has also been undertaken at Fittleworth Mill in 2016, which should improve that longitudinal connectivity within the lower catchment and reduce migratory costs (ARRT, 2017). This may improve access to the highly suitable spawning habitat created by riffle feature as part of the RHES further upstream. Further monitoring of this intervention is recommended to ensure access has been restored for all life stages.

Figure 6.19 Young fish (species unknown) using the fish refuges constructed in 2014 downstream of Fittleworth Mill.

Figure 6.20 Fenced fish refuge constructed in 2014 showing signs of infilling 1 year post-construction.

Cattle are well known geomorphic agents that contribute to bank erosion (Trimble and Mendel, 1995) and have been linked to the decline of brown trout (Summers et al., 2005). Riparian restoration through fencing can be successful in boosting the local populations of this species (Summers et al., 2008). In spring 2016, selective areas of fencing and tree planting were undertaken by the Arun and Rother Rivers Trust (ARRT) to mitigate areas of bank erosion along the channel between the study site and Fittleworth Mill. As with the other fencing undertaken in 2014 at the study site, in time this should provide additional shade, reduce the input of fine sediment into the system, and potentially enhance the quality of the existing marginal habitats. However, whilst targeted intervention in the areas restored by ARRT may prevent further erosion in these locations, the restoration (perhaps for financial constraints) does not account for the behaviour of cattle.

Cattle have been observed to spend up to 9% of their time in either the stream or the riparian zone, as they have a preference to defecate in-stream and occupy these environments during warmer temperatures (Bond et al., 2012). Therefore, without fencing the entire length of a reach, bank erosion is likely to re-occur in the areas which remain unfenced. This has been observed approximately 20m downstream of the study reach; poaching of the banks by cattle has created a small embayment (Fig. 6.21). This could be advantageous in creating a needed refuge habitat downstream of the riffle feature for the species evaluated in this thesis.

Figure 6.21 Bank erosion approximately 20m downstream of the study site in an unfenced section of the right bank observed in July 2016.

On the other hand, the addition of fine material into the channel is unlikely to benefit the wider ecological community, nor the operations of Southern Water abstraction site downstream at Hardham. Thus, if funding permits, fencing and refuge habitat creation are more likely to be effective and sustainable if carried out at a larger scale. Further monitoring would useful to identify if there is a marked difference in the rate of fine sediment delivery to the channel between either; the continued erosion of an established access point created by cattle, or by cattle developing a new access point following intervention. If piecemeal fencing along the channel were found ineffective an alternative could be to completely fence the entire channel (funding permitting) with the exception of some of the existing cattle access points. Ideally, these access points would be sited downstream of riffle features to provide shallow slow flowing physical habitats for refuge.

The results of this study have indicated that the riffle feature constructed as part of the RHES was resilient to significant in-channel flows (Chapter 5). The results of this monitoring suggest that the feature was particularly effective at promoting physical habitat heterogeneity and provisioning spawning physical habitats for brown trout, dace and roach at low flows. This would suggest that the riffle feature was (hydraulically) successful in achieving its objectives. However, this monitoring study also highlights the limited provision of physical habitat for juvenile and fry life stages within the reach and possibly the wider lower catchment (as the baseline morphology was largely representative of the channel in the lower catchment).

Recent restoration work undertaken within the lower catchment may have afforded some of that additional provision needed to help support younger fish through the life cycle. However, further monitoring and adaptive management of this environment are likely to be required to if this is to be successful. This is because deposition is the dominant geomorphological process within the lower catchment due to a very low slope, high sediment load and overwide geometry (Cox and Soar, 2017). The majority of recent restoration activities have focused on in-channel restoration and have not really addressed the catchment scale fine sediment input issues. These are thought to be a contributor to the poor ecological status of the river and sedimentation of gravels observed within the catchment (Evans et al., 2017).

Agriculture is an economically important activity within the catchment, therefore, fine sediment inputs from these sources may be difficult to significantly reduce. An evaluation of pilot sediment traps implemented within the catchment has indicated that the sediment traps may not be effective at preventing all fine sediment derived from agricultural fields from reaching the channel (Wright and Foster, 2014). Therefore, in-channel sediment management may be a critical factor in maintaining the geomorphological diversity and physical habitat provision created by the RHES. Particularly as recent sedimentological monitoring within the catchment has already indicated that the gravels of the riffle feature may be becoming saturated with fine sediment (Evans et al., 2017).

The overwide channel geometry and banks elevated above the floodplain are a legacy of the Rother Navigation. These features are likely to be contributing to fine sediment issues within the lower catchment by reducing floodplain connectivity (Cox and Soar, 2017) and also present a threat to species during higher flow events (Section 6.4). Therefore, the restoration of floodplain connectivity by reducing the bank height along the River Rother could be an effective and realistic management option. This may help to mitigate the impact of fine sediment pollution on spawning habitat within the catchment and improve the resilience of younger fish and less mobile species to high flow events.

The restoration of riparian meadows may also be beneficial for promoting floodplain sediment storage and bank stabilisation (Micheli and Kirchner, 2002a; 2002b). Meadows are natural buffer strips which may improve the water quality of surface run-off through filtering out nutrient rich sediment by up to 95% (Vought et al., 1995). Nationally, meadows have seen a rapid decline of 97% between 1930 and 1984 as a result of the intensification and mechanisation of agriculture (Fuller, 1987; Blackstock et al., 1999). It is possible that such an approach may have multiple ecosystem benefits by reducing flood risk within the wider River Arun catchment. This may be an important consideration as there is a significant flood risk posed to the lower River Arun (which includes Arundel and Littlehampton) now and in future with an estimated 1099 properties at risk by 2100 (EA, 2009).

At a more local scale, the overwide channel geometry may promote sediment deposition on the bed (Cox and Soar, 2017). Consequently, in-channel deflectors may be an effective measure to locally promote sediment transport to cleanse patches of gravel within the lower catchment of the River Rother. Additionally, gravel is unlikely to be naturally sourced to maintain the physical habitat afforded by the riffle feature within the lower catchment (Chapter 5). Therefore, regular gravel augmentation is recommended below Coultershaw Bridge (a significant in-channel barrier upstream of the study site) to replenish this feature. However, the consideration of the historical and present conditions of a catchment is critical in setting realistic restoration objectives (Wohl, 2005; Beller et al., 2016). Therefore, based on the review of catchment characteristics and processes in Chapter 3, the restoration of the lower catchment of the River Rother for brown trout is an ambitious target. If stakeholders are eventually successful in reaching this target, the physical habitat may require a continual high level of intervention to maintain it.

The results of this study suggest that physical habitat in the lower catchment of the River Rother may be more conducive (with some intervention) for supporting coarse fisheries. This is supported by EA fish surveys and catch reports from local angling clubs. Consequently, the restoration of physical habitat to support for coarse fish in the catchment may be a more realistic and economically sustainable target within the catchment. In 2005, the South-East of England had the most rod licence holders in the UK, 88% of time was spent on coarse fishing compared to 11% of time fishing for trout (Mawle and Peirson, 2009). Through angling activities, trout fisheries contribute ~£40 million per annum to the global economy, however, coarse fisheries contribute significantly more which is estimated at ~£130 million per annum (Mawle and Peirson, 2009). The restoration of the River Rother for coarse fisheries may be a more sustainable option for the River Rother, and resources for restoring trout fisheries may be a more effective and sustainable on other local chalk streams such as the River Meon or Itchen.

6.6 Key findings from the physical habitat performance evaluation of the RHES

The restoration of riffles is a widely used technique, although there are limited empirical studies on their performance to compare these results to. This study encourages the evaluation of similar schemes to improve on best practice within river management. This chapter has revealed some key findings which are of significance not only for the future management of the River Rother and similar environments but for the practice of wider river restoration monitoring design and monitoring. The salient findings can be summarised as follows:

- Undertaking baseline surveys and wider consideration of physical habitat provision are critical for identifying suitable river restoration objectives to support a sustainable ecological community. This study has highlighted that highly suitable physical habitat for a species may be inadvertently affected by restoration activities.
- The ADCP data can be used effectively when using supporting local and catchment knowledge to assess the physical habitat performance of a river restoration scheme. This additional information is critical to interpreting performance from monitoring data.
- Considering an appropriate spatial scale for the collection of physical habitat monitoring datasets is highly important, as they can influence the interpretation of physical habitat performance. This is study has highlighted that scale may be particularly important for the assessment of habitat heterogeneity.
- Different physical habitat assessments may afford different interpretations of performance. Therefore, a suite of methods may be used to provide an informed objective assessment. In addition, the extent of change during the monitoring period was not visually apparent, consequently rapid visual surveying techniques such as fixed-point photography and walkover surveys could have drawn different conclusions on the scheme's performance. Thus, this highlights the value of these data driven approaches, particularly where change may be visually obscured.
- Using spatial metrics in addition to summative methods (e.g. usable area) is important to assess the quality of physical habitats.
- Whilst in general habitat heterogeneity may be advantageous in supporting an ecological community, but in the design of pool-riffle sequences a more gradual bathymetric variation afford more preferential habitats for fish.

This chapter and Chapter 5 have demonstrated that the ADCP may be used to evaluate both geomorphological and physical habitat performance of a river restoration scheme. Chapter 7 will build on these results to evaluate the practicality of applying these high-resolution datasets collected using emerging technologies in practice beyond academic studies.

7 Science to Practice: Data-driven River Restoration Performance Assessments

7.1 Introduction

The River Rother Habitat Enhancement Scheme (RHES) monitoring programme afforded some valuable lessons of using Acoustic Doppler Current Profiler (ADCP). This monitoring programme demonstrates that high-resolution ADCP data may yield informative results for the adaptive management of river restoration schemes (Chapters 5 and 6). There is limited literature available on the use of the ADCP and similar technologies for this monitoring application. However, the literature suggests that these technologies have been used for research purposes to gain an improved understanding of the aquatic environment (Woodget et al., 2017). This chapter evaluates the value of high resolution data for river restoration monitoring (Section 7.2). The ADCP has found to be a reliable alternative to traditional surveying methods such as an Electromagnetic Current Meter (ECM) (Kinzli et al., 2011) except in extremely turbulent conditions (Neary et al., 2013; Richmond et al., 2014; Gunanwan et al., 2013). Therefore, an evaluation of the accuracy of this equipment is beyond the scope of this research. The chapter discusses the practical application of the ADCP and other technologies for routine data-driven performance assessments. Practical guidance is presented on what equipment and resolution is likely to be suitable given the catchment conditions of a scheme (Section 7.3.1).

Funding is a severe limitation of river restoration monitoring, particularly for longer-term more resource intensive programmes (Mainstone and Wheeldon, 2016). Therefore, the use of data driven performance assessments should be strategically undertaken to maximise the learning potential from river restoration projects with the resources available. This chapter presents guidance, based on the findings of this research, for allocating resources for river restoration monitoring (Section 7.4.1). A discussion on the practicality of tracking and disseminating project performance using data-driven performance assessments also features within this chapter. Data-driven performance assessments may not be so easily interpreted by non-technical stakeholders compared to other forms of monitoring (e.g. fixed-point photography). Therefore, this chapter discusses how data-driven performance assessments may be useful in the development of non-technical engagement tools (Section 7.3.1). There is a need for performance tracking tools for overall river restoration performance (Castillo et al., 2016). The concept of a performance tracking framework presented and evaluated in Section 7.4.2, this could integrate with existing learning platforms (Chapter 2). This tool is demonstrated using a hypothetical example based on the RHES data. The tool was not used to evaluate the performance of the RHES as suitable SMART objectives (a key component of the framework) were not set for this research project.

7.2 Data Resolution

Historically, physical habitat monitoring of river restoration schemes has been viewed as highly resource intensive (Maddock, 1999). Increasingly, new technologies have the potential to expedite the data collection process (Kinzli et al., 2011; Woodget et al., 2014; Woodget et al., 2016; Woodget et al., 2017). The resolution of in-channel data surveyed using traditional point-based measurement techniques is estimated as 10-15 cross sections per day in a 10 m wide channel (EA, 2007). Kinzli et al. (2011) estimated that an ADCP may be used to capture the same data as an Electromagnetic Current Meter (ECM) in 30% of the time. Therefore, new technologies such as the ADCP have the potential to capture significantly more data within the same amount of time as traditional surveying methods and improve data resolution. Alternatively, these technologies could be used to improve the length of the reach that could be monitored.

The data collected using the ADCP are spatially limited to the path of the equipment and are not as spatially continuous as data that could be collected using other technologies such as UAVs (Woodget et al., 2017). The ADCP has the potential to expedite data collection, but its use does not necessarily guarantee that data resolution will improve by applying this technology. The ADCP may be advantageous over other new technologies (such as UAVs) as it may be applied in poorer weather conditions, deep or turbid aquatic environments that photogrammetric methods may not be able to capture. It can still improve the speed of data-capture over traditional point measurement methods which may be beneficial in fast changing flow conditions or for health and safety reasons.

To demonstrate the effect of data resolution on the quality of river restoration monitoring results, the geomorphological and physical habitat changes between the immediate and 12-month post-construction surveys were evaluated at a lower resolution. These surveys were chosen as significant geomorphological and physical habitat changes were noted (Chapters 5 and 6). Therefore, it was interesting to evaluate whether a lower data resolution data set could effectively capture this change. These surveys were represented at a resolution that could have been realistically obtained within one day using point-based measurement techniques (e.g. Acoustic Doppler Velocimeter (ADV) and RTK GPS). The data were represented by sampling 10 cross-sections at regular intervals from the high-resolution data sets used in Chapters 5 and 6 (Fig. 7.1 & 7.2). These cross-sections were interpolated using same grid and other methods outlined in Chapter 4. This sampling technique assumed there were negligible errors in the data sets used in Chapters 5 and 6. Thus, this method assumed that the sampled velocity and depth values from these data sets were an accurate representation of reality.

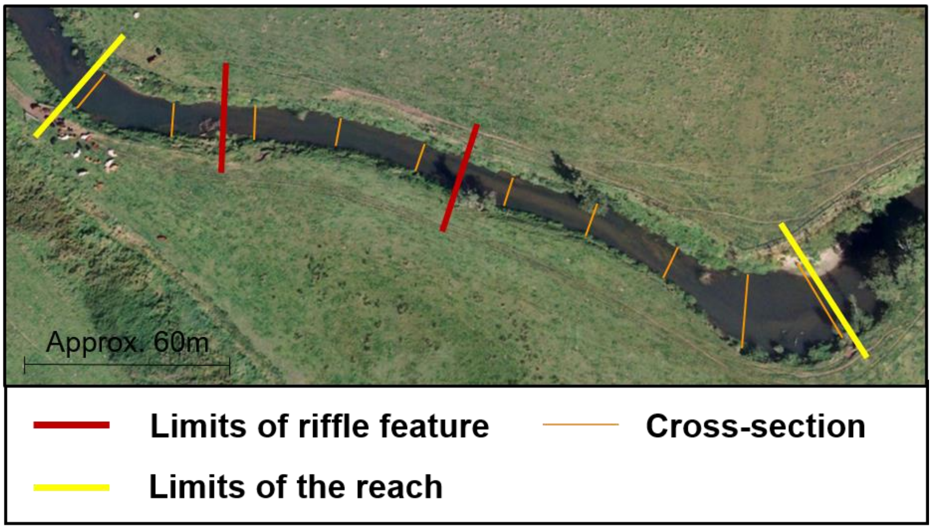


Figure 7.1 Approximate locations of the cross-sections sampled from the ADCP data used to evaluate the effect of resolution on the interpretation of physical habitat performance.

Scale can influence river restoration monitoring results and the ultimate interpretation of performance (Chapters 5 and 6). Consequently, maintaining the spatial scale of assessment was an important component of this evaluation. The interpolated data were subsequently evaluated using the same procedures outlined in Chapter 4 (unless stated otherwise below) to evaluate the benefit of using higher resolution data for assessing the geomorphological and physical habitat performance of river restoration schemes. The DEMs and physical habitat simulations derived from high and low-resolution data sets will be referred to herein as high and low resolution DEMs or data sets for simplicity. The high and low-resolution data were interpolated on to the grid (Chapter 4).

In addition to the methods outlined in Chapter 4, an analysis of DEM accuracy was performed to assess the error between the pre-interpolated data and the interpolated DEM (high-resolution data sets). This was achieved by subtracting the interpolated values from the pre-interpolated values. A negative value indicates the pre-interpolated elevation data was lower than the interpolated elevation data and a positive value indicates that the pre-interpolation elevation data was higher than the interpolated elevation data. The methods to analyse physical habitat performance also varied from those used previously within the thesis. The 40% depth velocity data was sampled from the high-resolution data set and used to simulate physical habitat suitability within this section. This was done to account for best practice point-velocity sampling methods which measure velocity at 40% of the depth above the bed to represent average conditions.

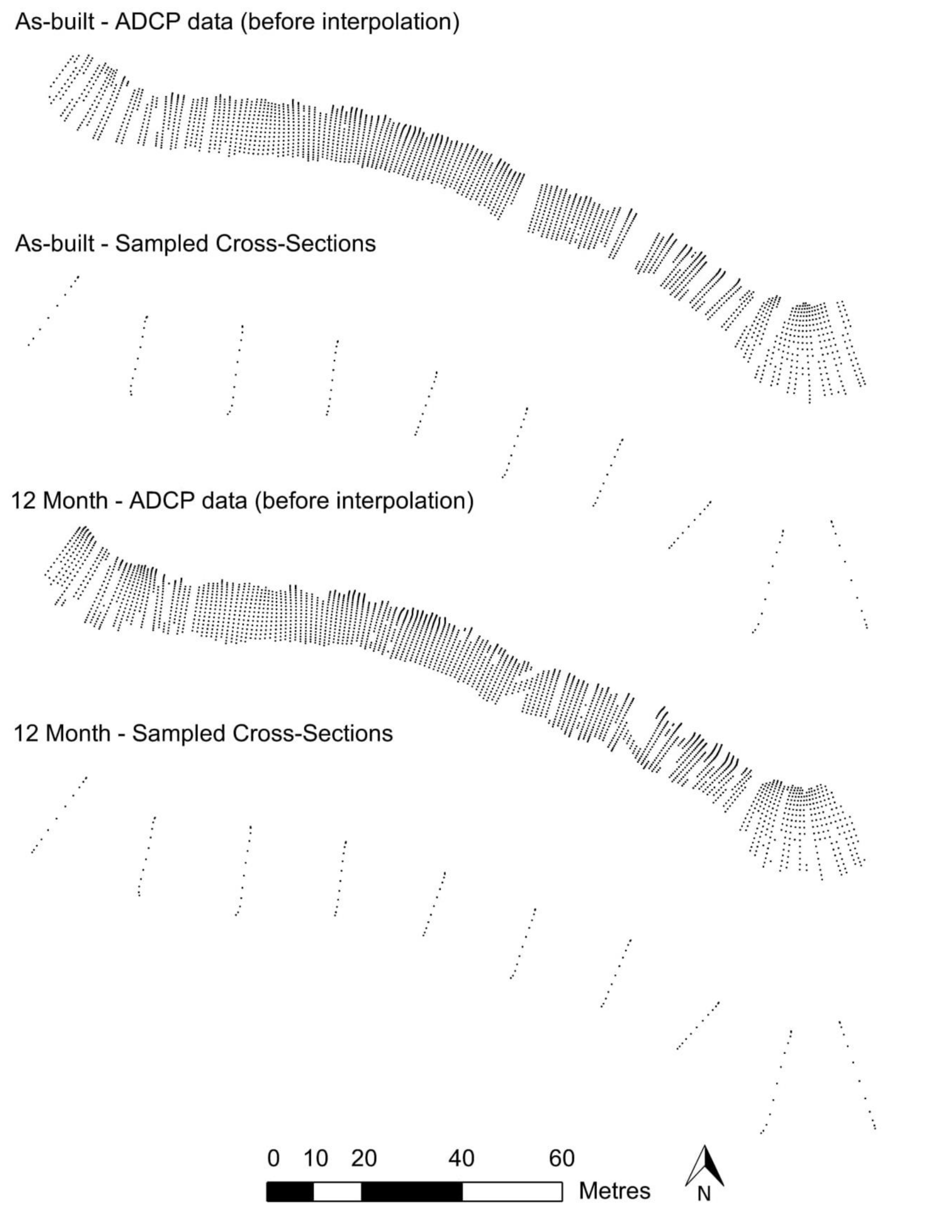


Figure 7.2 Data resolution of the high and low resolution as-built and 12-month surveys

In contrast, the physical habitat simulations of the RHES discussed in Chapter 6 utilised the improved data resolution in the vertical dimension afforded by the ADCP to give an estimate of average velocity. Therefore, the physical habitat simulation results reported within this section may vary to those in Chapter 6. This chapter explores the impact of using different velocity data on the simulation of physical habitat suitability. Using the high-resolution data, the effect of choosing different representations of a velocity on physical habitat provision was evaluated. Three representations of average velocity were explored;

- the average of 5 velocity points within the water column (0.2 Depth (D), 0.4 D, 0.6 D, 0.8 D and near surface),
- the average of 2 velocity points within the water column (0.2 D and 0.8 D), and
- 0.4 D velocity point.

A fourth velocity point of 0.2D was also used to evaluate the effect of evaluating the suitability of physical habitat using average conditions, when a species may in fact use only certain areas of the water column (e.g. near the surface or bed).

7.2.1 Geomorphological performance

The analyses of the low-resolution elevation data sets of the as-built and 12-month post construction surveys (August 2013 and August 2014, respectively) indicated similar bed morphologies to those observed from the high-resolution data sets (Fig. 7.3). The analysis of the as-built survey data sets indicated that there were two distinct morphological forms within the reach, these were; an area of higher elevation where the riffle feature was constructed, and an area of lower elevation typical of a deep pool in the downstream end of the reach (Fig. 7.4). The analysis of the 12-month post construction survey data sets indicated that there were three distinct morphological forms within the reach. These were an area of higher elevation were the riffle feature was constructed and two areas of lower elevation typical of a deep pool at either end of the reach (Fig. 7.4). These forms were less well defined in the DEMs from both surveys derived from the low-resolution data set as transitional areas between forms were smoothed during interpolation. Additionally, as to be expected, much of the micro-scale topographic variation was lost.

The variation in the DEMs of different resolutions was reflected in the moving window analyses. The mean elevations of both the as-built and 12-months surveys were higher over the downstream pool and lower over the riffle feature when compared to the high-resolution data (Fig. 7.4). This is possibly the result of the interpolation process between data points. The variation between estimated elevations from the high and low-resolution data sets was particularly noticeable over the riffle tail during the 12-month post construction survey, with a difference of ~20cm (Fig. 7.4). Whilst the overall features may be detected using lower resolution data, these results indicate that low resolution data cannot represent form as accurately as high-resolution data. The quartile coefficient of dispersion (QCD) values also displayed similar trends in general yet exhibited more variability between the DEMs derived from high and low-resolution data sets. The QCD values from the low-resolution DEM analyses indicated that pools had a less varied morphology than suggested from the high-resolution DEM analyses (Fig. 7.4). This is likely to be a function of the more gradual morphologies represented by the interpolation of the low-resolution datasets.

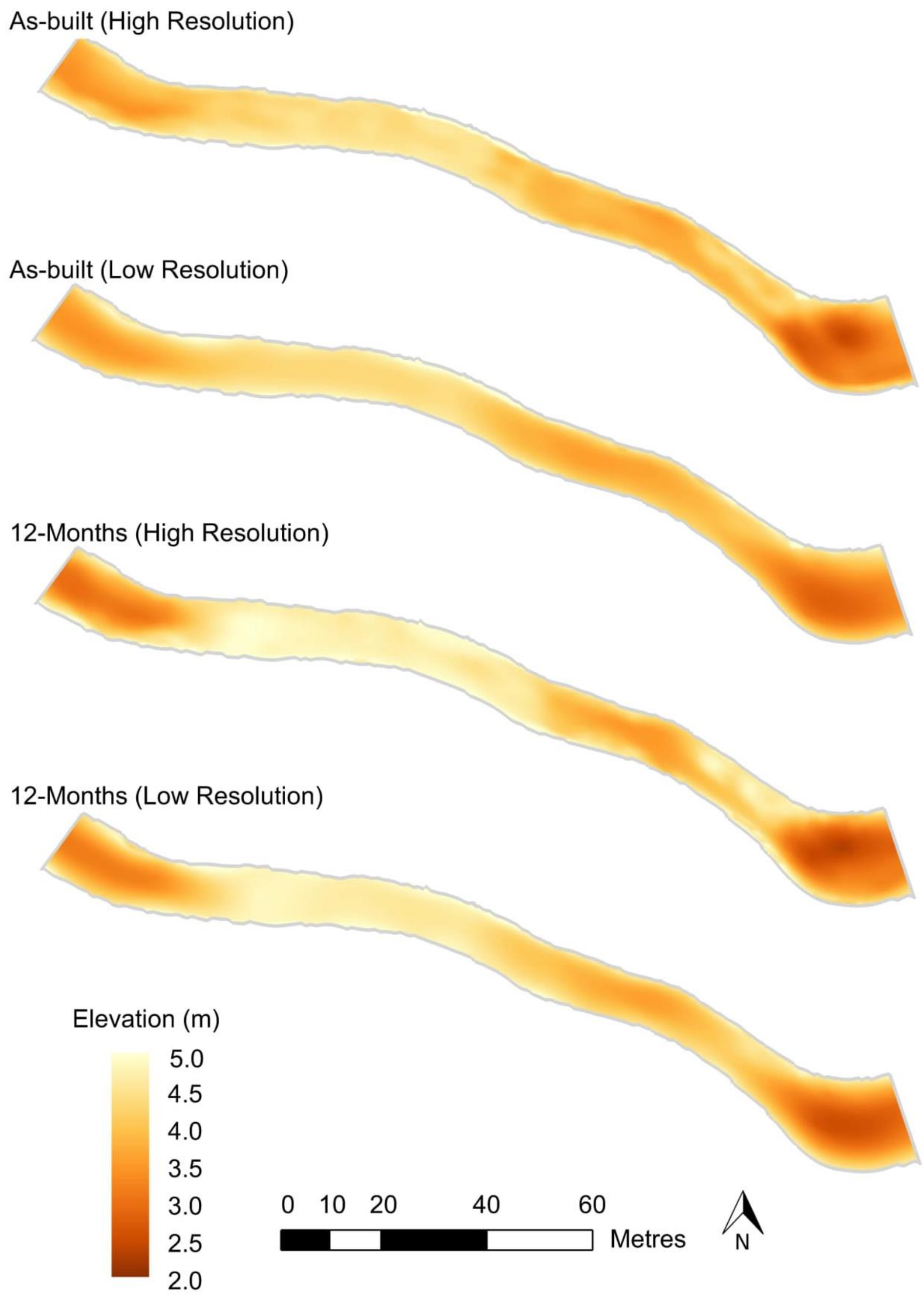


Figure 7.3 DEMs of the as-built and 12-months post construction survey derived from both the high and low-resolution elevation data sets.

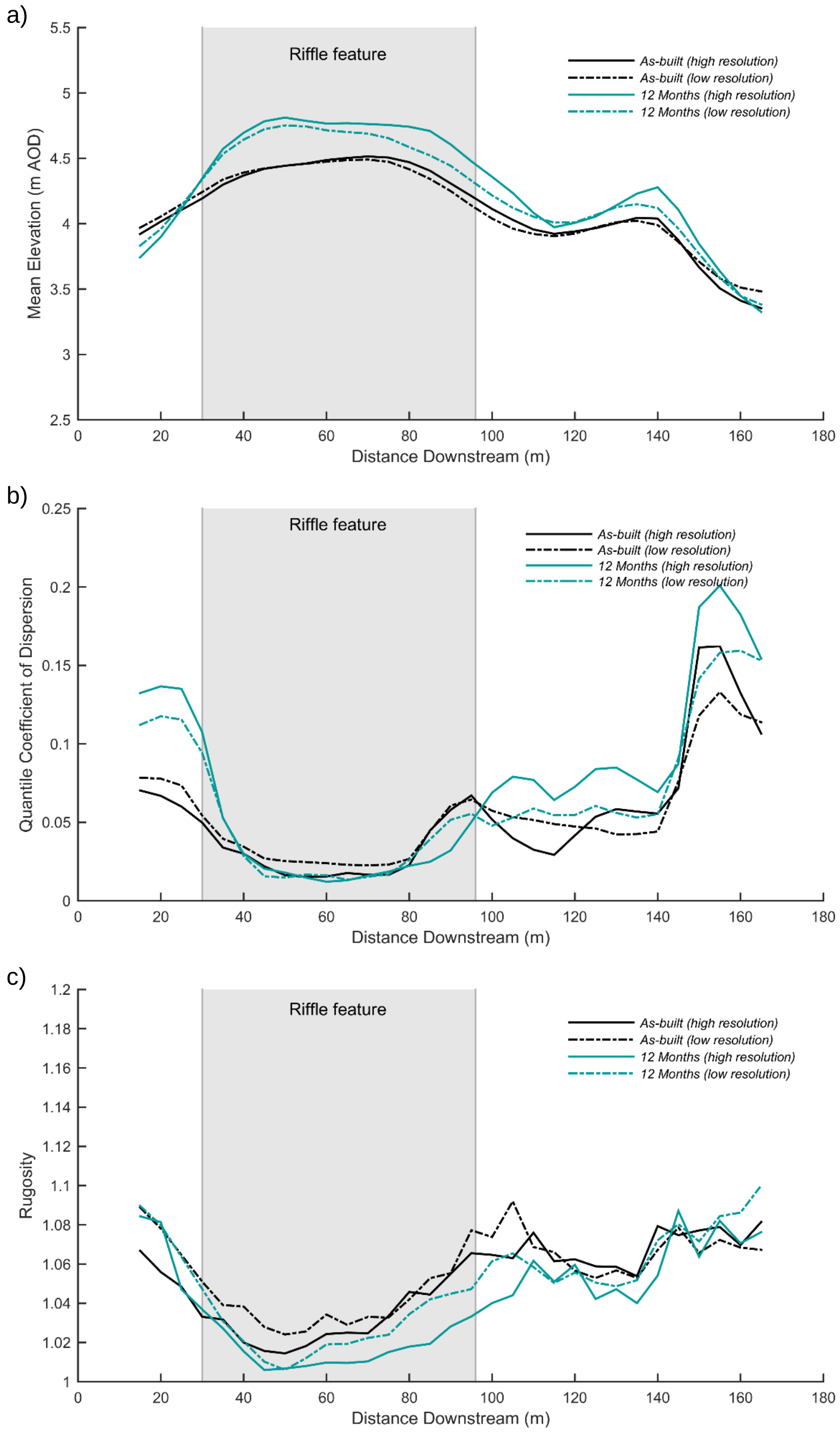


Figure 7.4 Moving window analysis of the both the low and high-resolution elevation data sets of the as-built and 12-month post construction surveys.

The more gradual morphologies represented by the low-resolution DEMs also appeared to influence the rugosity values (an indicator of topographic complexity) (Fig. 7.4). The moving window analysis of rugosity suggested that the high-resolution DEMs were generally slightly less topographically complex. Overall, the high-resolution DEMs presented more distinct morphological forms and less substantive transitional forms. However, the low resolution DEMs suggested transitional forms were more prominent. Thus, the high-resolution DEM exhibited forms which were more uniform at the local scale, and over-representation of transitional areas could suggest that the low-resolution DEMs were more topographically complex at the local scale. The broad trends in the moving window analysis of rugosity were similar between the high and low-resolution data sets.

The high-resolution DEMs indicated that the structural integrity of the constructed riffle feature was compromised as sediments were re-worked during the flood events and the crest of the riffle feature was re-sited over the head of the riffle feature (Chapter 5). This change was detected in the 12-month post-construction survey, along with significant scour in the upstream and downstream pools within the reach. The low-resolution DEMs indicate a similar trend and detected these major geomorphological changes (Fig. 7.5). Smaller scale geomorphological changes (such as localised scour downstream of the riffle feature) were not detected from the lower-resolution data-sets (Fig. 7.5).

Table 7.1 Geomorphological change estimates between the as-built and 12-month surveys of different resolutions.

Data Resolution comparison	Total Volume Change (m^3)	Net Volume Change (m^3)	Percentage Volume of Scour (%)	Percentage Volume of Fill (%)
Low – Low	380.2	125.8	33.5	66.5
High – High	549.6	213.5	30.6	69.4
Low – High	636.3	184.9	35.5	64.5
High – Low	443.2	154.4	32.6	67.4

Volume change estimates from the lower-resolution data sets were of a lower magnitude than estimated by the higher resolution data set (Table 7.1). For example, the high-resolution data sets indicated a net gain of 213.5 m^3 of sediment but the low-resolution data set indicated a significantly lower estimate of 125.8 m^3 of sediment (~41% less). It is possible that by sampling at a lower resolution, smaller geomorphological adjustments (that cumulatively may account for a significant proportion of change) may be undetected. Interestingly in this example, despite the variability in the magnitude of change, the ratio of scour to fill was similar (Table 7.1). This suggests that whilst the comparison of low-resolution data sets may not necessarily give an accurate estimate of geomorphological change, their comparison may be indicative of the broader geomorphological processes following restoration.

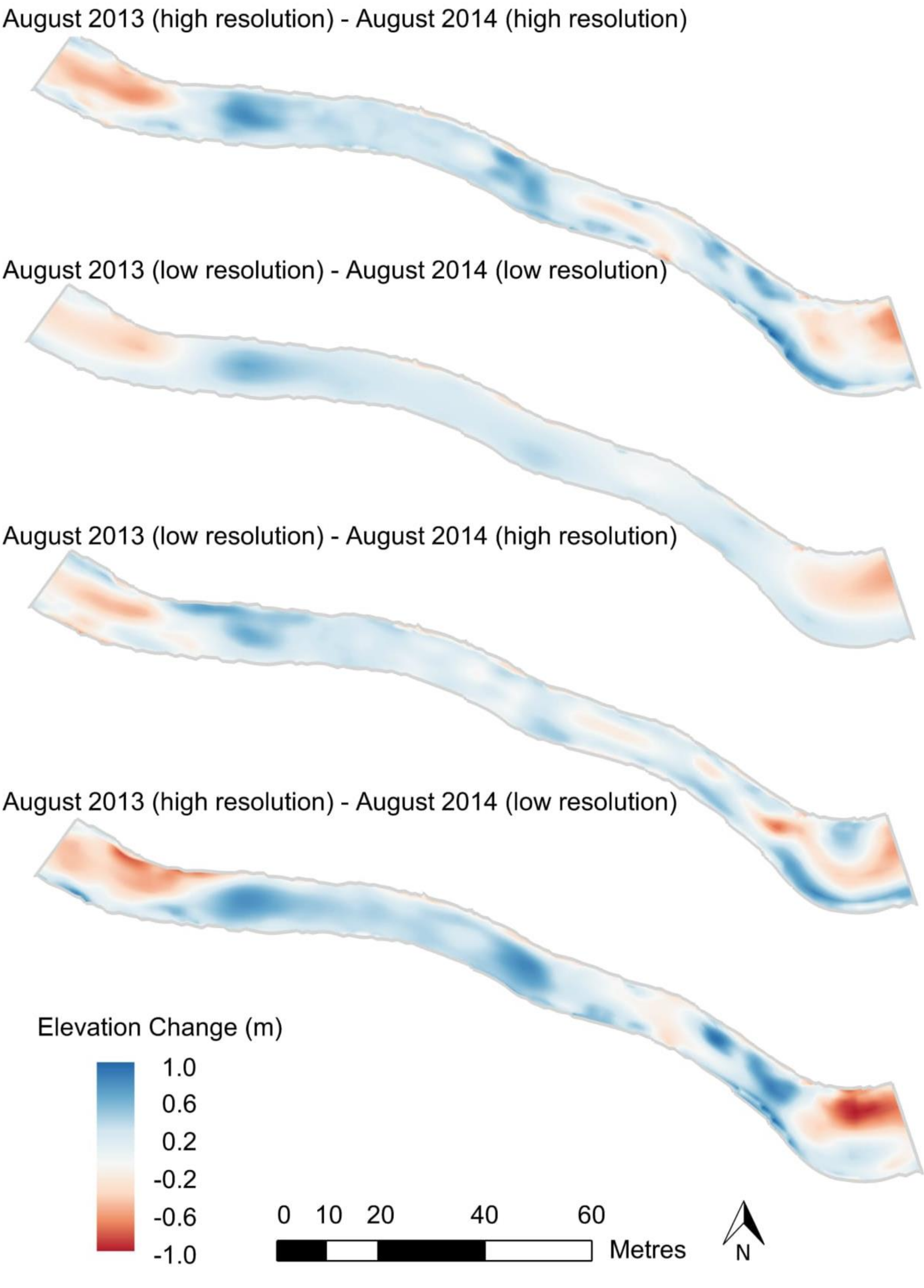


Figure 7.5 DEM of Difference maps between the as-built and 12 months post construction survey using different resolutions of data.

The comparison of high and low-resolution data sets from different surveys also gave variable results to the high-high resolution interpretation. The comparison of an as-built low-resolution data set and a 12 month high-resolution data set estimated that ~14% less material had been gained within the reach than when using both high-resolution data sets. The comparison of an as-built high-resolution data set and a 12 month low-resolution data set suggested ~28% less material had been gained when using high-resolution data sets. The comparison of high and low-resolution data sets may give closer approximations of geomorphological change estimates than low-low resolution comparisons. However, the spatial representation of these changes (Figure 7.5) were significantly varied and could be misleading in the interpretation of geomorphological processes. Therefore, maintaining the resolution of data when comparing different surveys and different restoration schemes would be preferable.

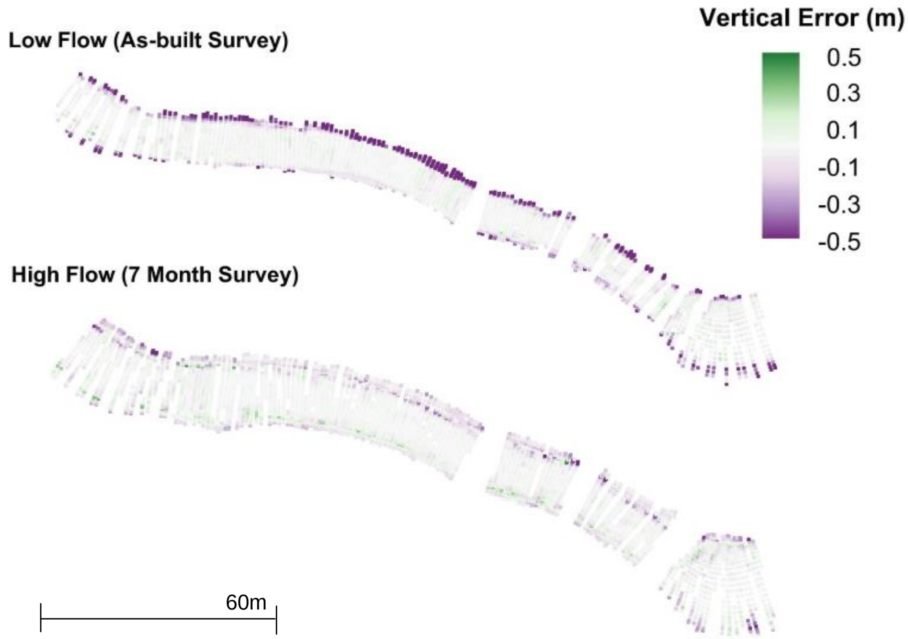


Figure 7.6 DEM error analysis, positive values indicate the pre-interpolated data was higher than the DEM and negative values indicate that the pre-interpolated data was lower than the DEM.

An analysis of high resolution DEM accuracy indicates that the error between the pre-interpolated data and the interpolated DEM was greatest in the channel margins (Figure 7.6). Larger errors were observed in the margins of high resolution DEMs derived from data captured during low flow conditions, a potential result of seasonal vegetation restricting data collection in the margins. However, some errors were also observed in the margins of the high resolution DEMs using in data captured during higher flows. This may suggest that low-resolution floodplain data could have also introduced error into the DEM.

Thus, it is recommended here that high-resolution in-channel data is complemented with a higher resolution terrestrial data set (collected using either Terrestrial Laser Scanning (TLS), Structure from Motion (SFM) or open LiDAR data) to reduce the introduction of error in margins during interpolation of the data. An accurate representation of the marginal areas is important for river restoration monitoring as these areas provide valuable refuge habitat (Chapter 6).

Insights for data-driven geomorphological performance assessments

This research has indicated that high-resolution data collection may afford a more accurate representation of geomorphological forms and micro-scale topographic complexity. Therefore, high-resolution, or even better, spatially continuous data would be a preferable option for river restoration monitoring in situations where there are no resource constraints. In particular, higher resolution data may be especially useful for studies that require more accurate estimations of geomorphological form and change. For example, schemes that are planned within an urban environment, near to critical infrastructure or sites of cultural heritage, where significant geomorphological change may be undesirable.

This research has highlighted that geomorphological forms and change may not be visually detectable and therefore in some situations fixed-point-photography and rapid visual assessment methods may not be appropriate. Thus, whilst data-driven performance assessments may be more resource intensive they can also add significant value to monitoring programmes in some environments. The exploration of data resolution has also highlighted that low-resolution data (as defined by EA (2007)) may provide a less accurate but still adequate interpretation of river restoration performance. Thus, in some situations (for example where resources are limited or flow conditions change rapidly), it may be important to sacrifice data resolution to maximise the spatial extent of the monitoring programme as scale can be very influential on monitoring results (Chapters 5 and 6).

This evaluation demonstrates the importance of maintaining consistency in data resolution between data sets to derive accurate representations of geomorphological change. Comparing the analyses of different resolution data sets resulted in spurious spatial patterns of change. This is an important observation as it may be realistic to assume that multiple pieces of equipment may be used between surveys within a monitoring programme. For example, operational limitations may restrict the use of certain equipment during some surveys. Also, new technologies may be introduced within a monitoring programme, particularly over a longer-term monitoring programme. Moreover, the equipment available to different schemes will almost certainly vary due to differing levels of resources and expertise available. Consequently, if data resolution varies between surveys or schemes, consideration should be given to resampling the higher resolution data sets to match the resolution of the lower resolution data sets used to derive DEMs.

Based on this observation, it is recommended that consistency is sought between data sets where possible and that baseline data sets are monitored at the highest spatial resolution practical. If a monitoring programme lacks adequate baseline data sets, the value of post-project data sets are much lower as there are no comparative surveys to measure success. There are very few higher resolution monitoring data sets of restoration schemes widely available at present for comparative analysis (Chapter 5). Consequently, if a scheme is pioneering in terms of a novel design or application of a design to a new environment, higher resolution monitoring is also recommended at this stage in river restoration practice. Over time, if resource constraints decrease, data-driven performance assessments may become the norm with in river restoration practice, and these data sets may be more directly comparable.

Clearly, improved spatial data resolution may improve the accuracy of geomorphological monitoring results. However, given the resources are available, the improved efficiency from new technologies could facilitate an improvement in temporal data resolution. Best-practice guidance suggests that topographic surveying should be undertaken following implementation, 3-months post-construction to capture immediate change, and thereafter annually to capture any further geomorphic change. It also suggests that additional surveying is undertaken following any high flow events (EA, 2007). The RHES monitoring results support this guidance. A near bankfull flow event occurred prior to the 2013/14 flood events, data captured following this event was critical in evaluating the performance the riffle feature as constructed. Similarly, the data collected immediately following the 2013/14 flood events and 12-months post-construction enabled an assessment of the riffle feature's resilience. These surveys also highlighted sediment transport processes that were potentially ecologically relevant.

The significant flow event of the 2013/14 floods was quite obviously a channel forming event on the River Rother. However, for other schemes, the magnitude of flow events may not be apparent, and there is a need for guidance on which flows signify the need for further monitoring. Whilst this may be assumed to be a bankfull flow, channel forming flows are dependent on a variety of variables including catchment geology and substrate, climate and human modifications. A substantial amount of research on the effective discharge has been undertaken (Soar and Thorne; 2011), but this is an area that warrants further research particularly to inform long-term river restoration monitoring practice.

The effective discharges will likely be calculated during the design phase of the river restoration scheme to determine a stable design within minimal risks of erosion or sedimentation (Doyle et al., 2007). However, if the design, construction and monitoring phases of river restoration are undertaken by different stakeholders, this information may not be widely communicated. Therefore, it is recommended that where possible stakeholders partake in all phases of river restoration monitoring to promote knowledge exchange. Additionally, further development of guidance on this may be beneficial to improve understanding of river restoration performance and resilience. For now, at least, event based resurveying is likely to only be undertaken following very significant flow events due to limited resources. This practice will undoubtedly yield valuable learning outcomes, yet the knowledge on the performance of schemes with respect to smaller but still effective flows may not be fully established.

This research also suggests that when undertaking only an annual topographic resurveying strategy, seasonal patterns may be overlooked. The 18-month survey was not required according to best-practice (EA, 2007), but the analyses using this data set identified seasonal scour and fill patterns within the reach which are critical for adaptive management. These potential seasonal topographic changes have implications for physical habitat quality provision throughout the year, but also reveal insights into changes which occur seasonally and not just as a function of high flow events. Therefore, the identification of these processes may be important to incorporate into future restoration design. Again, at least for the foreseeable future, the temporal resolution of surveys is likely to be dictated by resources. The author made recommendations to stakeholders that the monitoring of the RHES continued, yet an immediate response from a stakeholder was 'who will pay for it'. Additional to the constraint of resources, is the willingness and ability of stakeholders to commit over longer timescales. Pearson et al. (in prep) highlight that historically, communities had a vested interest in managing the landscape but this may no longer be as important to local communities. Consequently, given this lack of social connectivity within river systems it may be challenging to maintain the enthusiasm for data collection in the long-term.

7.2.2 Physical habitat performance

The impact of data resolution on depth and depth-average velocity may also have implications for the interpretation of physical habitat performance. The assessment of the Hydromorphological Index of Diversity (HMID) using low resolution data suggested a more homogenous habitat for both surveys (at the reach and riffle scale) than when using the high-resolution data. This observation may be attributed to the loss of detail on smaller-scale in-channel complexity. However, the overall changes in form were still detected when using low-resolution data. The high-resolution data indicated no substantial change in habitat heterogeneity at the reach-scale following the 2013/14 floods, whereas the low-resolution data indicated that habitat became more homogenous. The broad trends were still detected from the low-resolution data set, but the variation between the HMID values over both scales (reach and riffle) was much more notable when using different velocity representations at high flows. The HMID values using the 5-point average velocity data (high-resolution) indicated the riffle feature presented a more homogenous habitat in both surveys than the HMID values using the 40% depth velocity data (high resolution). The HMID values using the 5-point average velocity data suggested a significant improvement in habitat heterogeneity over the reach following the 2013/14 floods. However, the HMID values calculated using the 40% depth velocity data indicated no substantial variation in habitat heterogeneity had occurred.

Table 7.2 Variability in HMID values dependant on the resolution of the data and representation.

	HMID High resolution (5-point average velocity)	HMID High resolution (40% depth point velocity)	HMID Low resolution (40% depth point velocity)
As-built survey			
Reach	8.6	9.2	8.5
Riffle	4.9	6.5	6.1
12-month survey			
Reach	10.8	9.2	7.5
Riffle	4.5	6.4	5.5

A similar broad trend of physical habitat performance (i.e. improvement or deterioration evaluated using suitability criteria) of the RHES for each species was observed from both the high and low-resolution data sets (Table 7.3). However, the interpreted degree of improvement or deterioration to physical habitat performance was varied (Table 7.3). For example, during the as-built survey 112 m^2 less physical habitat suitable for adult roach was estimated when using low-resolution data rather than high-resolution data (Appendix C). Conversely, during the 12-month post-construction survey 112 m^2 more physical habitat suitable for spawning brown trout was estimated when using low-resolution data rather than high-resolution data (Appendix C). As a result, the degree of improvement was interpreted as more significant when estimated by the low-resolution data than the high-resolution data (Fig. 7.7, Appendix C).

Table 7.3 The comparison of physical habitat performance interpretations from high and low-resolution data sets. The symbols imply physical habitat performance (PHP) has either seen a ➔ slight improvement, ➔ significant improvement, ➔ slight deterioration or ➔ significant deterioration or = no change.

	As-built-12 months (high-high resolution)	As-built-12 months (low-low resolution)	Change in PHP interpretation by using low resolution data
Brown Trout			
Adult	↑	⬆	More improvement
Juvenile	↓	↑	More improvement
Fry	↑	↑	No change
Spawning	⬆	⬆	No change
Dace			
Adult	↑	⬆	More improvement
Juvenile	⬇	↓	More deterioration
Fry	=	=	No change
Spawning	⬆	↑	Less improvement
Roach			
Adult	↑	⬆	More improvement
Juvenile	↑	⬆	More improvement
Fry	=	=	No change
Spawning	↑	⬆	More improvement

In another example, during the as-built survey, 104 m^2 less physical habitat suitable for spawning brown trout was estimated when using low rather than high-resolution data. Similarly, during the 12 month post-construction survey, 76 m^2 less physical habitat suitable for spawning brown trout was estimated when using low rather than high-resolution data. Again, in this example, the change in suitable physical habitat area for brown trout was also greater when estimated by the low-resolution data. However, the differences in the spatial configuration of this physical habitat between the high and the low-resolution surveys were more pronounced (Fig. 7.8).

Observations of the physical habitat provision for some species suggest that the low-resolution data may be less effective for detecting the fragmentation of physical habitat. This is not necessarily surprising as a level of detail is lost when using low resolution data. The edges of physical habitat simulated using the low-resolution surveys were much smoother and had a lower edge ratio than simulated using the high-resolution surveys. In landscape ecology, patches with more complex shapes (i.e. a greater edge ratio) are more likely to be subject to biotic and abiotic disturbances (Fagan et al., 1999). The edge ratio has been used within this project as a measure to assess fragmentation. Edge effects have not been discussed in detail within this thesis as the literature available relevant to aquatic environments appears to be limited to marine environments (Jelbert et al., 2010; Smith et al., 2008). Nonetheless, it highlights an important component of habitat assessment that high-resolution data may be able to inform in the future, subject to further research within freshwater environments.

The effect of low-resolution data sets on the interpretation of physical habitat performance can be illustrated by the simulation results of juvenile dace. The high-resolution data set suggested that the highly suitable habitat for juvenile dace became more fragmented at low flows following the 2013/14 flood events, but the low and moderate suitability physical habitats became less fragmented (Fig. 7.9, Appendix C). In contrast, the low-resolution data set suggested that the low, moderate and high suitability physical habitats all became more fragmented following the 2013/14 flood events. Furthermore, the degree of fragmentation of the highly suitable habitat was underestimated by all of the fragmentation metrics. This is an example of how fragmentation is likely to be underestimated when using low resolution data.

This example highlights that fragmentation metrics within the assessment of physical habitat performance may provide an extra level of detail that can help interpret the quality of physical habitat provision. The law of parsimony states that the best option is normally the simplest, thus, raising the question of whether this extra assessment of fragmentation affords ‘added value’ or ‘added complexity’. To evaluate this, and assess the value of high-resolution data, the performance of the physical habitat results reported in Chapter 6 were re-interpreted without the consideration of fragmentation metrics and based solely on the abundance of habitat. Approximately half of the physical habitat simulations were interpreted differently (Table 7.4), with the majority of these viewed more favourably.

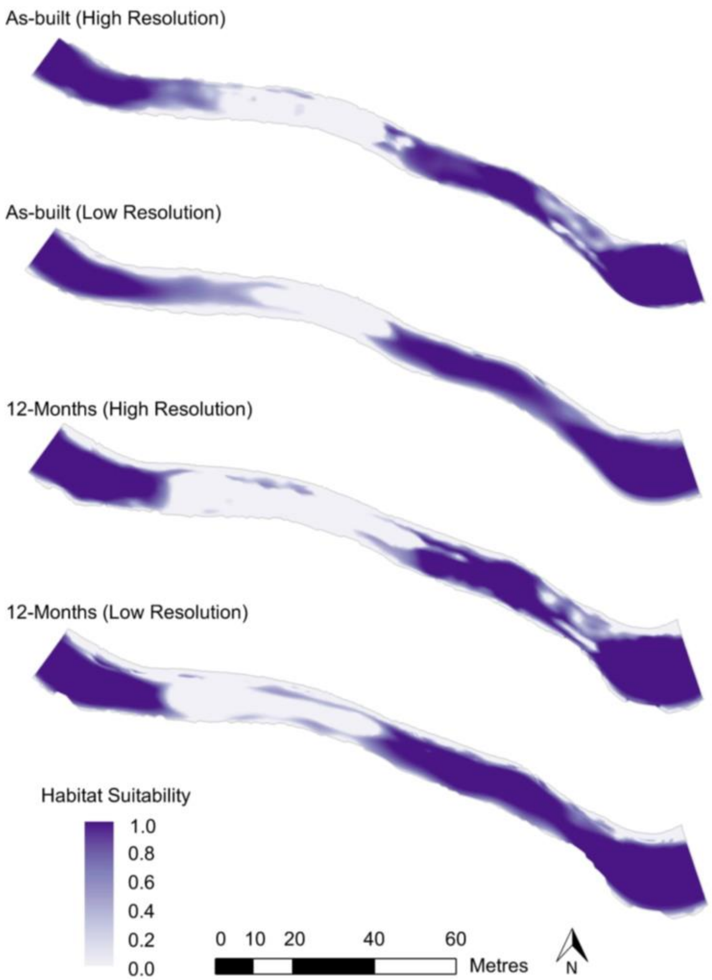


Figure 7.7 Physical habitat suitability for adult roach over a 180m reach of the River Rother near Shopham Bridge using high and low-resolution data sets of the as-built and 12-month post-construction survey

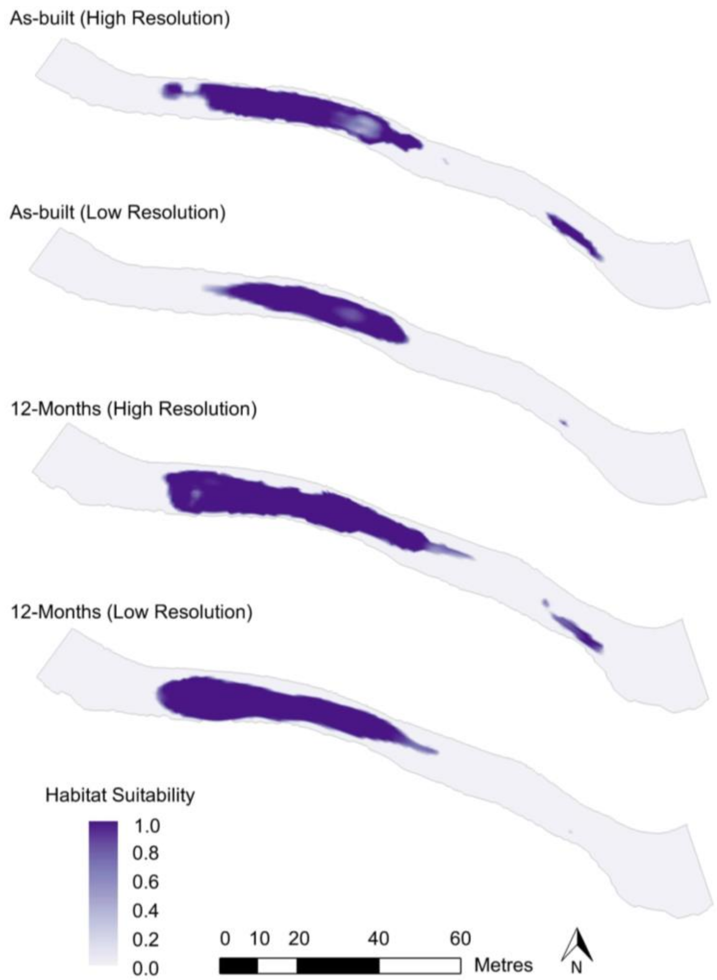


Figure 7.8 Physical habitat suitability for spawning brown trout over an 180m reach of the River Rother near Shopham Bridge using high and low-resolution data sets of the as-built and 12-month post-construction survey

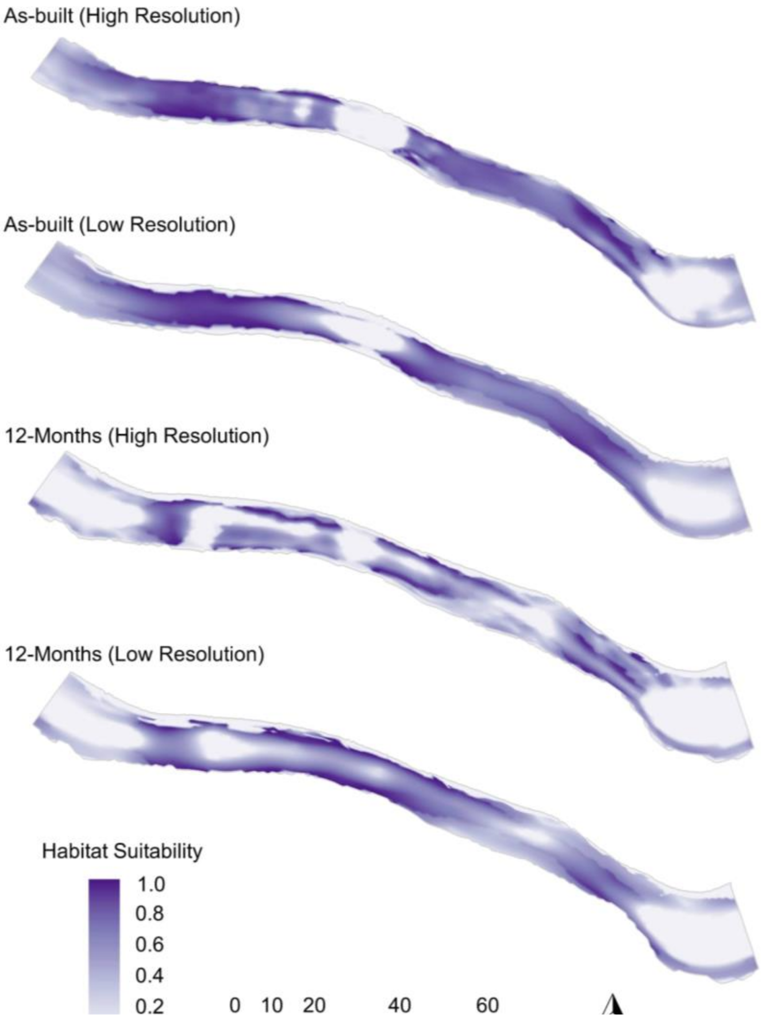


Figure 7.9 Physical habitat suitability for spawning brown trout over a 180m reach of the River Rother near Shopham Bridge using high and low-resolution data sets of the as-built and 12-month post-construction survey

Interestingly, the physical habitat simulations that were interpreted differently did not seem to be associated with the shape of the preference curves for the species. Instead, they appeared to be closely associated with species that were adequately provisioned for prior to restoration. Therefore, the use of fragmentation metrics and high-resolution data may be particularly beneficial for evaluating the physical habitat of species which may be more vulnerable to fragmentation. This highlights the importance of baseline monitoring for identifying high-quality physical habitats which may be at risk by the implementation of restoration measures. If these high-quality physical habitats are desirable, high-resolution data collection may be a preferred option for monitoring post-project physical habitat performance. If the deterioration of this habitat is unlikely to be detrimental to the population of the species (e.g. found abundantly elsewhere as exemplified in Chapter 6), low-resolution data collection may be adequate for monitoring physical habitat performance.

Table 7.4 Impact on implied of physical habitat performance by not including an assessment of spatial configuration of physical habitats. The size and the direction of the arrows indicate the scale of improvement or deterioration of physical habitat provision within the time periods, as seen in Chapter 6. The symbol colour demonstrates how the interpretation of habitat provision can be altered by not including spatial configuration metrics in an assessment; when using global metrics only (e.g. WUA) ***grey*** *indicates no significant change in interpretation,* ***yellow*** *indicates a positive interpretation of physical habitat provision and* ***black*** *indicates a negative interpretation of physical habitat provision. The asterisks denote the comparison between surveys as either * comparable low flows ** low to moderate flow comparison, *** comparable moderate flows **** moderate to high flow comparison.*

	Habitat Heterogeneity Assessment		**Physical Habitat Simulation**											
			Brown Trout				Roach				Dace			
	HMID	HCA	ADULT	JUVENILE	FRY	SPAWNING	ADULT	JUVENILE	FRY	SPAWNING	ADULT	JUVENILE	FRY	SPAWNING
Baseline to as-built *	↑ (grey)	↑ (grey)	↑ (grey)	↑ (grey)	↑ (grey)	↑ (grey)	↓ (yellow)	↓ (yellow)	↑ (grey)	↑ (grey)	= (grey)	↓ (yellow)	↑ (grey)	↑ (grey)
Baseline to 12 Months *	↑ (grey)	↑ (black)	↑ (black)	=	↑ (yellow)	↑ (grey)	↑ (yellow)	↑ (yellow)	↑ (grey)	↑ (grey)	↑ (yellow)	= (yellow)	↑ (yellow)	↑ (grey)
As-built to 12 Months *	↑ (grey)	=	=	↑ (yellow)	↑ (yellow)	↑ (grey)	↑ (yellow)	↑ (yellow)	=	↑ (grey)	↑ (yellow)	↑ (yellow)	↑ (grey)	↑ (grey)
As-built to 3 Months **	↓ (grey)	=	↑ (grey)	↑ (yellow)	↓ (grey)	↑ (yellow)	↑ (yellow)	↑ (yellow)	↓ (grey)	↑ (grey)	↑ (yellow)	↓ (grey)	↓ (grey)	↓ (grey)
12 Months to 18 Months **	↓ (grey)	=	↑ (grey)	↑ (yellow)	↓ (black)	↑ (yellow)	↑ (yellow)	↑ (yellow)	↓ (grey)	↑ (yellow)	↑ (yellow)	↓ (black)	↓ (black)	↓ (grey)
3 Months to 18 Months ***	↑ (grey)	=	↓ (grey)	= (grey)	= (grey)	↑ (grey)	=	=	= (grey)	↓ (grey)	= (yellow)	↓ (black)	=	↑ (yellow)
18 Months to 7 Months ****	↓ (grey)	↑ (grey)	↑ (yellow)	↓ (grey)	↑ (yellow)	↓ (grey)	↑ (yellow)	↑ (yellow)	= (grey)	↑ (grey)	↑ (yellow)	↓ (black)	=	↓ (black)

Whether the improved resolution of velocity measurements in the vertical dimension captured by the ADCP adds value or complexity to the physical habitat assessment was also assessed. In the example of spawning brown trout, the target species and life stage of the RHES, the choice of velocity significantly altered the results of the physical habitat simulation. The averages of 5 and 2 velocity points within the water column afforded very similar spatial patterns and abundance (399 m^2) of physical habitat throughout the reach (Fig. 7.10; Appendix D). However, physical habitat quality was lower when using the average of 5 velocity points within the water column. Ecological validation of suitability criteria could potentially allow the development of guidance on choosing appropriate velocity measurements for physical habitat assessments. Consequently, the improved vertical resolution of velocity measurements may add significant value in providing more accurate physical habitat assessments.

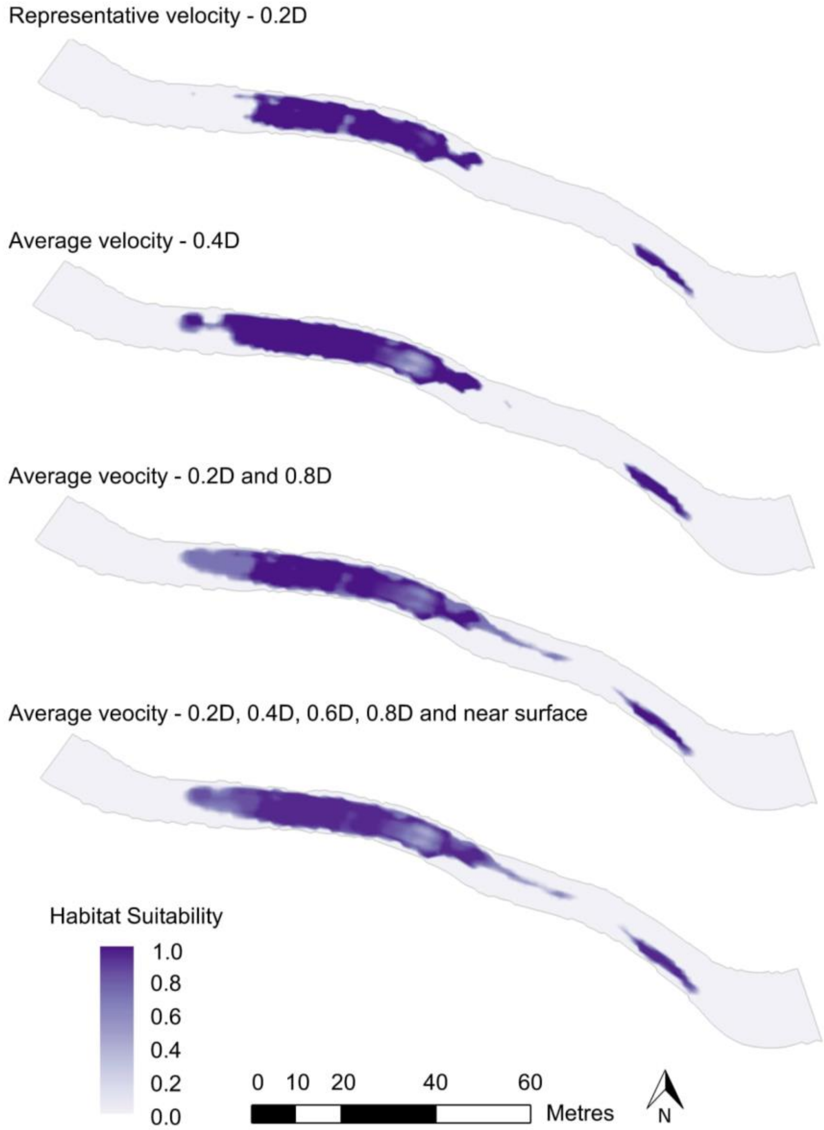


Figure 7.10 Physical habitat suitability for spawning brown trout over a 180m reach of the River Rother near Shopham Bridge in August 2013, using different representations of velocity from the water column.

Insights for data-driven physical habitat performance assessment

The RHES monitoring programme has provided some insights that may have wider implications for data collection guidance for river restoration schemes. The resolution of physical habitat data appeared to have a significant effect on physical habitat assessment results, both habitat heterogeneity and physical habitat suitability results. Therefore, high-resolution data collection surveys are recommended to improve the accuracy of physical habitat performance assessments. However, it was still possible to detect the general trends of physical habitat performance from low resolution data sets. It is unlikely (at least for now) that higher resolution data collection will be routinely undertaken for every river restoration monitoring programme due to resource constraints. Therefore, the observations from this research may help direct resources for higher resolution data collection to certain schemes.

In terms of physical habitat suitability assessments, data resolution affected both the estimation of total suitable area and fragmentation. Higher resolution data collection was superior for detecting physical habitat fragmentation. This was most evident in species which were adequately provisioned for prior to restoration, but intervention resulted in a decline in physical habitat provision. It was also evident where physical habitat was provisioned adequately at a single discharge, but changes to discharge resulted in a decline in physical habitat provision. Consequently, high resolution data collection is likely to be more beneficial in these situations. These situations may be identified through higher resolution baseline monitoring and design modelling. Therefore, this research recommends that baseline monitoring should be conducted at the highest resolution possible to help target the resources of river restoration monitoring effectively.

The learning outcomes from the RHES were significantly limited by a lack of temporal baseline data (Chapters 5 and 6). Although a comprehensive baseline survey is presented in this thesis, this data set captured only the pre-restoration condition at a low flow immediately prior to restoration due to a short lead in time. The baseline survey in this thesis has provided useful insights into the performance of the scheme with respect to low flow physical habitat provision and morphological diversity. However, additional baseline data sets at a range of flows would have been beneficial in the interpretation of seasonal and long-term restoration responses. This would have maximised the learning outcomes of the RHES performance. Furthermore, the collection and analysis of data could have informed the design of the RHES and highlighted the need to focus on fry as well spawning habitats for fish species within the reach.

The importance of baseline monitoring is strongly reiterated within peer-reviewed literature (e.g. England et al. (2008); Brierley et al. (2010)) and practical guidance (e.g. RRC, 2011). However, guidance on the frequency and duration of baseline monitoring is lacking. Given the recurrence interval for a bankfull flow event is estimated as 1-2 years, 2 years could be considered as the minimum period for baseline monitoring to maximise the chance of capturing a high flow event prior to restoration. A comprehensive baseline monitoring data set would also be beneficial for detecting geomorphological change. However, the results of this research have suggested flows exceeding bankfull may be more significant in influencing the morphology of some rivers. Such flows are likely to have a longer return interval and are unlikely to be captured within the pre-project monitoring period. Therefore, hydraulic modelling would be useful to estimate the impact of these flows and thus support the design of river restoration schemes.

The recommendations for geomorphological performance assessments following this research suggest that it may be important to sacrifice data resolution to maximise the spatial extent of the monitoring programme. This is also the case for physical habitat performance assessments, and highlighted through the HMID assessment. The ADCP is very versatile as it may be used to capture data at either a high or low resolution. However, the value of the ADCP in improving data resolution in the vertical dimension was highlighted by both the HMID assessment and physical habitat simulations. The results of both methods were significantly affected by the representation of velocity. This highlights the importance of choosing a representation of velocity appropriate for the aims and objectives of the restoration scheme. However, there is a need to develop a better understanding of the ecological relevance of velocity variation within the water column, to support informed decision making of choosing a representative velocity.

7.3 ADCP for data-driven performance assessments

This research has demonstrated that the ADCP may be used as a suitable alternative to other more efficient methods of higher-resolution data collection (e.g. photogrammetry) if the use of these alternatives is impractical or infeasible, as suggested by Marteau et al. (2016). Furthermore, this research has indicated that the ADCP may be advantageous over other technologies as it can improve the spatial resolution of velocity in the vertical dimension as well as horizontal (discussed in the previous section). The ADCP has been widely applied for the academic study of aquatic environments (e.g. Jamieson et al., 2011; Williams et al., 2015), yet not routinely for river restoration monitoring. The RHES monitoring programme afforded an excellent opportunity to explore the practical aspects of collecting data using the ADCP. These insights are valuable for understanding how the use of this technology could be developed such that it may be applied more routinely outside of academia.

7.3.1 Data Collection

As the ADCP has been used routinely for streamflow measurement, many models and their graphical user interfaces (GUI) of operating software have been refined. Consequently, the ADCP is very easy to operate, even for non-technical experts. Most of the data collection process is automated and basic training (less than one day) would be required to learn how to collect suitable data for geomorphological and physical habitat assessments. The ADCP may therefore be advantageous over other novel data collection technologies that may require significant pre-operational training (e.g. flight training for UAVs). A significant benefit of the ADCP over traditional technologies is the reduced health and safety risks to the operator. It would not usually require the operator to enter the channel. Therefore, the ADCP presents the opportunity to capture and analyse river restoration performance at greater range of flows and in more dangerous environments (e.g. river restoration involving weir removal). However, as with similar technologies (e.g. UAVs, Woodget et al. (2017)), the battery life (particularly of the remote device used to operate the equipment) was limited which could potentially disrupt surveying.

A significant proportion of surveying time during this study was used to manoeuvre around two trees within the reach. Consequently, established riparian vegetation, in-channel features and/or other significant barriers on the banks could reduce the efficiency of the data collection process when using the ADCP. This is could be an issue when using ropes to manually transverse the ADCP across the channel, especially during higher flows when the wetted channel width typically increases. Autonomous platforms such as the ‘Arc Boat’ (HR Wallingford, 2014) or the ‘Jetyak’ (Sherwood, 2013) and mounting of the ADCP onto small vessels, such as kayaks (Water Cube, 2016) could be used in these situations. However, these platform solutions are usually costlier which could reduce the feasibility of using an ADCP for river restoration monitoring. Additionally, these platforms are generally larger in size which may limit the measurable area of the cross-section, thus reducing the feasibility of using an ADCP for river restoration monitoring in smaller streams. Further research has focused on the development of the ‘Arc Lite’, a smaller version of the autonomous Arc Boat (Everard, 2014). This would be highly beneficial for river restoration monitoring for the reasons discussed above, although this is still likely to be costlier than a non-autonomous solution.

The restoration of lowland rivers and urban rivers are more common as they are more likely to have been previously modified, and there may also be a greater community pressure to restore them (Smith, Clifford and Mant, 2014). These rivers may be more turbid due to run-off particularly during higher flow events (Lawler et al., 2006), thus limiting the application of some data collection methods (e.g. SFM) for river restoration monitoring. The RHES, for example, was undertaken in a lowland sandy catchment with high water turbidity even at low flows. Consequently, SFM would not have been a feasible option for this environment and the ADCP was a useful asset for river restoration monitoring.

The ADCP appeared to be a very versatile tool for river restoration monitoring that could be deployed at short notice. Consequently, the potential for improving the understanding the restoration of different environments at a range of flows may increase. The ability to react quickly may be particularly beneficial for undertaking river restoration monitoring in catchments with a flashy flood hydrograph. The ADCP may be a good tool for reactive monitoring and would, for example, be more suitable than a UAV, which may require more extensive permissions. For example, in urban environments permission from the Civil Aviation Authority (CAA) for flying UAVs for commercial work is required (CAA, 2017). However, it is worth noting that multipath in urban environments could introduce error into the ADCP GPS data (Rennie and Rainville, 2006). This may be resolved by using an RTK system, but this may have cost implications.

The ADCP is not a panacea for geomorphological and physical habitat data collection, but it is an option for increasing data resolution particularly in environments where other cost and time efficient methods for higher resolution monitoring (e.g. UAV-SFM) may not be appropriate. For example, the ADCP would be advantageous over SFM in turbid or deep environments, as this method cannot currently collect data in these environments (Woodget et al., 2014). Additionally, the ADCP could be a more suitable option for monitoring during poor weather conditions, heavily tree-lined channels or in populated areas where the use of an UAV may be limited. Ultimately, the technology used for river restoration monitoring will largely depend on the state and availability of the equipment, the environment in which it will be deployed and the funding available to support the monitoring project. Based on the experiences of this study, data collection using the ADCP was very practical and is unlikely to be a barrier for applying this technology beyond research institutions for geomorphological and physical habitat performance monitoring. Guidance is presented based on these experiences in Figure 7.11 to highlight when an ADCP, UAV or point measurement technologies may be beneficial for collecting hydraulic physical habitat variables. In reality, the choice of technology will be affected by resource constraints, which are a significant barrier to the feasibility of data-driven performance assessments and will be explored in Section 7.4.

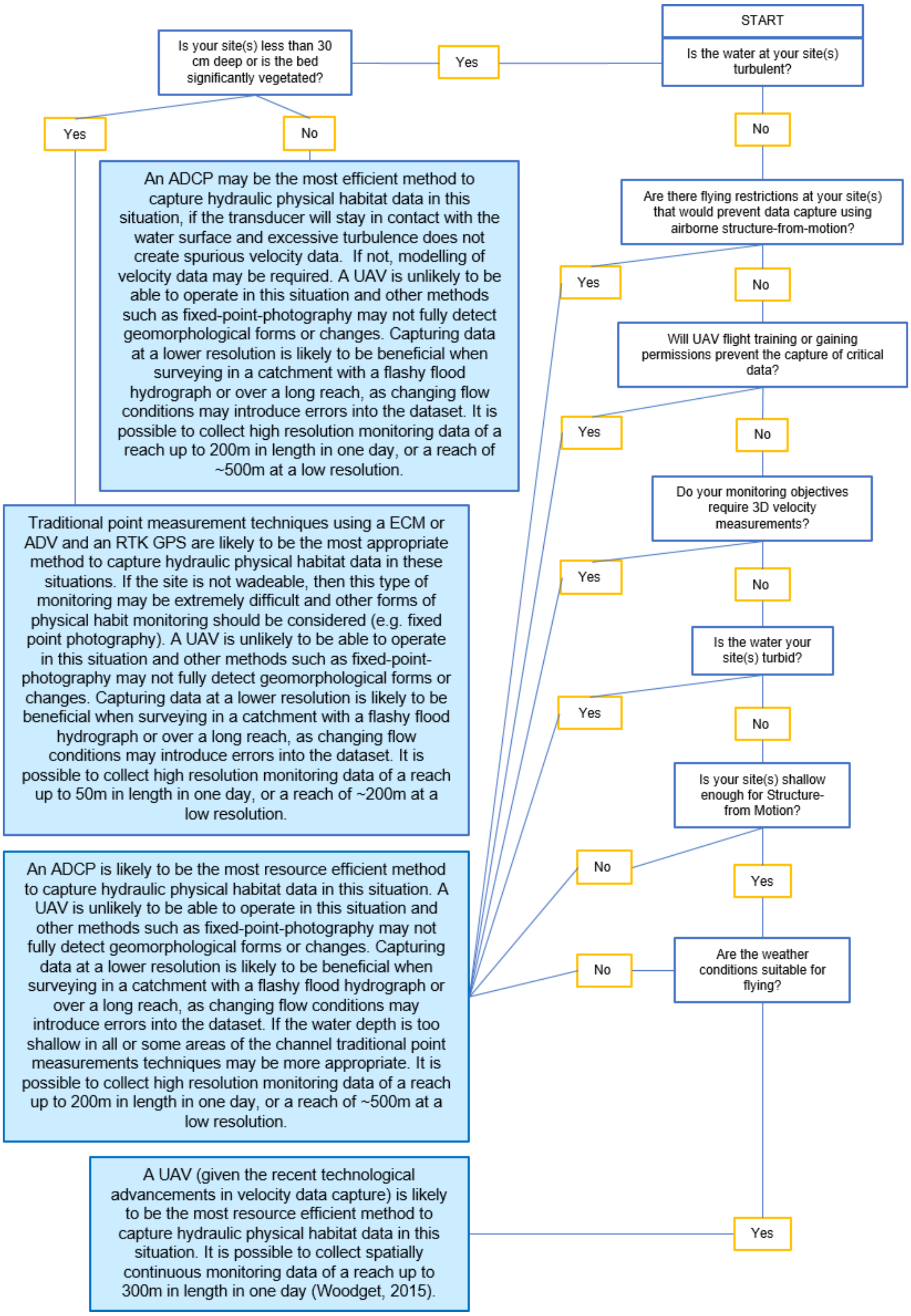


Figure 7.11 Guidance for choosing appropriate data collection methods for a river restoration monitoring survey. This assumes no resource constraints. A UAV (given the recent advancements technological advancements in capturing velocity data) is soon likely to be the most efficient method for physical habitat monitoring. However, there are environmental conditions where the ADCP may be more appropriate and these are outlined.

7.3.2 Data Analysis

The exploratory monitoring programme of the RHES highlighted the versatility of the ADCP data for learning from river restoration performance. The two different approaches to spatial analyses used in this thesis (Chapter 4.3) have highlighted the impact of scale on the interpretation of physical habitat performance when using high resolution ADCP data. For example, following restoration the assessment of habitat heterogeneity over the reach improved, thus indicating an improved ecological performance. However, if just considering the area over the riffle feature, the habitat appeared to become more homogenous, indicating that scheme had not performed as well. The moving window technique, however, yielded interesting findings on the structural complexity of forms within the reach which may have otherwise been missed. Therefore, this research highlights the importance of analysing physical habitat at a range of scales, even within reach scale studies.

This research also highlights the benefit of using different analyses to assess physical habitat performance when working with high-resolution ADCP data, as different methods can afford different interpretations. For example, habitat heterogeneity indicators afforded varying interpretations of the effect of the riffle feature on diversifying habitat at a range of flows. The hydromorphological index of diversity indicated that the riffle feature became less effective at diversifying habitat as discharge increased, whereas hydraulic clustering analyses indicated that hydraulic patches became more ecologically favourable as discharge increased. Furthermore, this research supports Carnie et al., (2015) in that global abundance metrics such as weighted usable area can be misleading in the interpretation of physical habitat provision. Therefore, the use of reachscape configuration metrics alongside physical habitat modelling when using ADCP data may be highly beneficial to the assessment of physical habitat performance of river restoration schemes. The interpretation of these reachscape configuration metrics is, however, largely subjective and a future research focus should be to validate these metrics with ecological data.

The ADCP data has been used in this research to monitor the geomorphological and physical habitat performance of a river restoration scheme at a high resolution. The data resolution and expediency of the RHES monitoring surveys was almost certainly an improvement on traditional point surveying methods. However, significant post-processing of the data was required prior to analysis (Section 4.2). Prior to this research project, the ADCP had primary been used for routine streamflow measurement and not for the detailed analysis of bathymetry and 3D-velocity measurements. Since a protocol was developed in this research, the post-processing of further ADCP data for other restoration schemes should take significantly less time. Nonetheless, at present, a minimum of 2-3 days in addition to data collection would be required to process the data from a survey and a minimum of 2-3 days for the subsequent analysis.

The operation of the ADCP generally requires little training but the post-processing and analyses of the data would require a set of specific skills, either gained by training or experience. Therefore, additional training (of a week as a minimum) may be required to enable others to replicate this work on other restoration schemes. Alternatively, data processing and analysis could be performed by an external commercial organisation which would remove the knowledge and skills requirement of practitioners. For example, WaterCube Inc. is a company in the US that offer a service to post-process data and provide software to visualise the outputs. However, both the post-processing service and cube-it software license require a fee (D. Krupa, personal communication, 14th December 2015). This could have further financial implications for routine river restoration monitoring. Consequently, without further development of post-processing and analysis software, the routine application of the ADCP to river restoration monitoring may be hindered due to resource constraints. Based on these observations it is not wholly surprising that the ADCP for this type of application has mostly been applied by research institutions where specialist skills sets and resource constraints may be more adequately provisioned.

A future avenue for development of this research study would be to automate the data-processing and analysis protocol (Chapter 4) as far as possible. This may alleviate resource implications associated with the post-processing of ADCP data and data from other technologies. In addition, the use of alternative open-source software packages to those used in the data analysis for this research (e.g. Scilab (Scilab Enterprises, 2015) instead of Matlab (Mathworks, 2014)) would increase the accessibility and feasibility of using such technologies for more routine river restoration monitoring. However, some open source software packages are sometimes less powerful and stable which may add processing time to a monitoring project. Additionally, some ADCP manufacturers appear to prefer compatibility with Matlab. For example, RiverSurveyor Live (Sontek equipment software) currently exports to Matlab.

In the long-term there is a need to go beyond publishing code and focus on the development of research software so academic research is fully repeatable (Hutton et al., 2016; Hut et al., 2017). This development of suitable software may be important if re-sampling and analysis of a higher resolution survey may be required. This research has highlighted this may be required for a more accurate comparison between different resolution surveys. Without at least some automation of data processing and analysis this may have significant resource implications, and thus may become unfeasible.

Resource constraints on monitoring, particularly in the UK, has led to the increasing involvement of citizen science within river restoration monitoring. The development of accessible research software that may be operated by non-experts (eg. community volunteers or undergraduate dissertation students) with a small amount of training (1 day for data collection and analysis combined), would be highly beneficial. The interpretation of results may still need some expert involvement but this research software may significantly reduce resource constraints. Pearson et al., (in prep) highlight that community involvement may be critical in river management. The benefits of utilising the local universities and communities for river restoration monitoring may be twofold; for maintaining engagement, but also reducing resource constraints. This may improve the likelihood of river restoration monitoring occurring and increase its longevity. In summary, this study has emphasised that both the resolution and novel analysis of ADCP data may be beneficial for river restoration practice. However, the post-processing and analysis could present significant barriers to the techniques used within this thesis outside of academia. Therefore, this should be a priority for further development.

7.3.3 Data Dissemination

At present, the outcomes of high-resolution data analysis, due to their novelty, are prime material for peer-reviewed journals and academic conferences. However, these mediums may be physically inaccessible and they may also require a certain level of expertise to assimilate the findings. Some peer-reviewed journals such as Earth Surface Processes and Landforms do not accept case-studies without 'wider systematic relevance' (Wiley, 2017). Consequently, the salient findings of a scheme may be obscured and harder to interpret by a non-expert if this is the only method of dissemination used. Publishing data sets along with journal articles is encouraged but not always necessary, therefore, access to data sets of previous schemes for comparative analyses may be limited. Conceivably, a one-stop repository for data, reports, articles and guidance etc. relating to each scheme may be beneficial to facilitate learning within river management. The RiverWiki presents an excellent example of this and is increasingly being used internationally. The further advocation of this resource is recommended.

The results of the RHES monitoring programme were disseminated at a range of different events to practitioners, academics and non-experts, including international, national and regional conferences as well as stakeholder meetings. At these events, the most excitement and discussion was generated over the planform representations of geomorphological change and physical habitat simulations, rather than graphs or values. Consequently, planform visualisations of the ADCP data have been used extensively within this thesis. This positive reaction to these visualisations amongst a range of stakeholders may be because they are recognisable as a river form. Graphs, values and even cross-sections are potentially less accessible to those who have not been involved within the project and to non-experts.

These visualisations may be incorporated into almost all forms of dissemination for river restoration monitoring outcomes, including posters and presentations at conferences, peer-reviewed journal articles, reports, social media, online databases. It is feasible, that the generation of these visualisations could be mostly automated given further development of research software. Furthermore, the 3D visualisation of these data sets may be even more relatable and could be combined with other forms of monitoring such as fixed-point-photography (Fig. 7.12). The data sets may potentially be used to develop augmented reality dissemination tools at a relatively low cost (C. Skinner, personal communication, 24 February 2016). Examples of similar tools include Humber-in-a-box and Flash Flood! that have presented opportunities to disseminate research findings on flooding (Skinner and van Rij, 2015; SeriousGeoGames, 2017). The development of similar tools for river restoration could be easily transportable to a variety of events.

Alternatively, these resources could be made available at river restoration sites through Wi-Fi hotspots which allow stakeholders to access resources and visualise performance in-situ. For example, archaeological research was disseminated at Peworth Park, West Sussex using Wi-Fi hotspots to run the ‘Interactive Park Explorer’ (National Trust, 2017). The author is not aware of river restoration monitoring data being widely disseminated using similar methods. These opportunities present areas for future development as they have the potential to improve accessibility through removing technical jargon and engage stakeholders with both the design and project performance of river restoration schemes. Figure 7.13 demonstrates that high resolution data may be better to develop representations of channel forms for augmented reality resources than low resolution data. Augmented reality dissemination tools may be a prime use for data-driven river restoration monitoring data. Schemes which may have a high profile, strong local opposition, or a strong community engagement component are considered a high priority for data-driven monitoring in this thesis. However, it may be difficult to assimilate the key findings from such visualisations and to gauge the overall level of success of a scheme. There is scope for a novel method which integrates data-driven monitoring outputs within other modes of dissemination (such as guidance literature, databases and social media) to non-technically track performance. This will be explored in Section 7.4.

In summary, the findings of Sections 7.2 and 7.3 (Figure 7.14) indicate that the ADCP may be an effective tool for collecting data for data-driven performance assessments. Comprehensive baseline data sets are a core component of these assessments and further promotion of the value of capturing these data sets is recommended. ADCP data is very versatile and may be analysed using a variety of methods to understand the geomorphological and physical habitat performance of river restoration schemes. However, the post-processing and analysis of data from innovative technologies, such as the ADCP, are highly resource intensive. The development of research software would alleviate some of the resource constraints associated with data-driven performance assessments. Additionally, the ADCP data has the potential to inform both technical and non-technical stakeholders of physical habitat performance. However, it may be difficult to easily establish and track project performance. Section 7.4 will explore some recommendations to improve the practicality of learning from data-driven performance assessments.

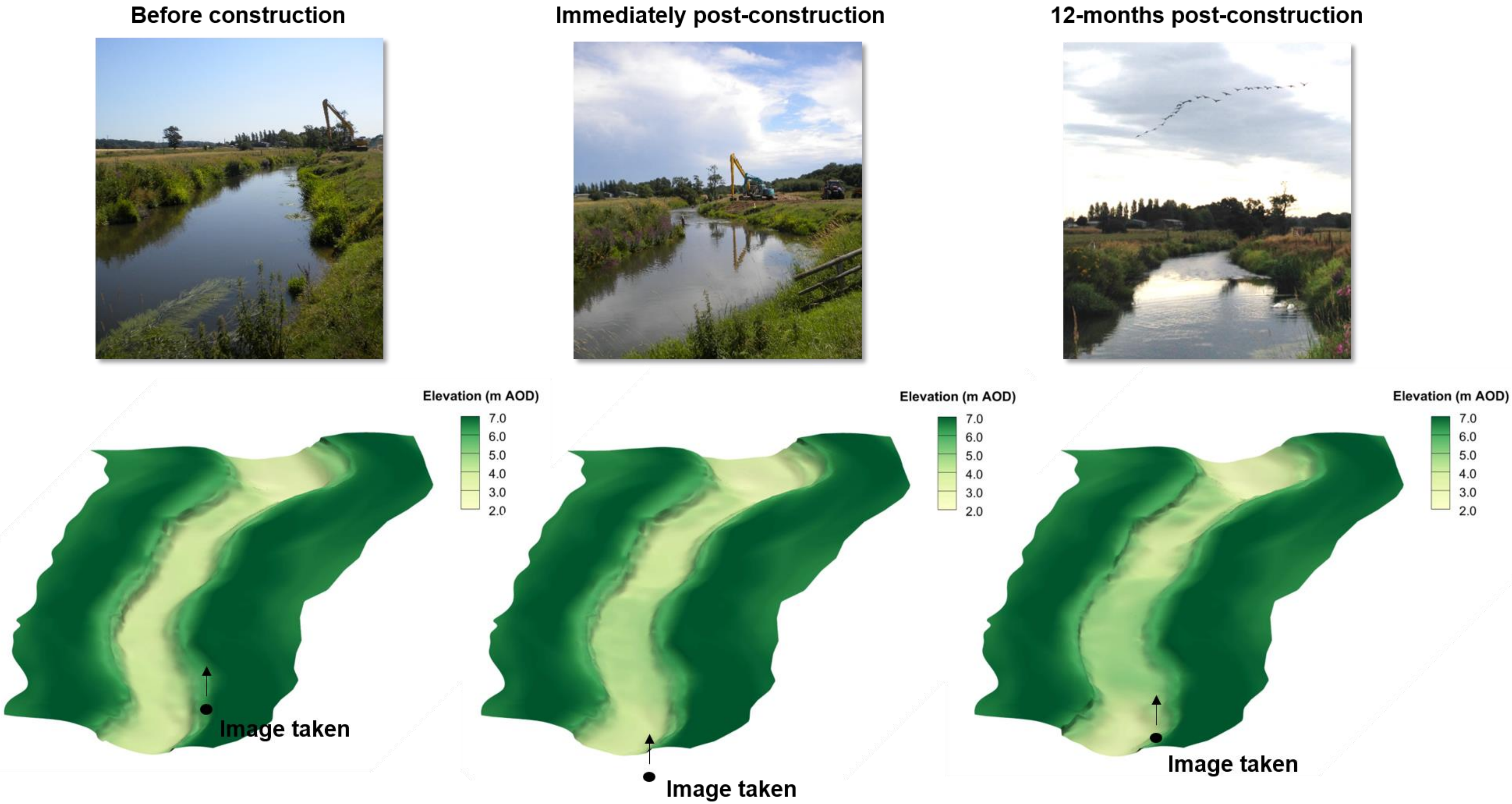


Figure 7.12 Demonstration of 3D visualisation of DEMs derived from high resolution data used with photography to potentially disseminate findings of the RHES

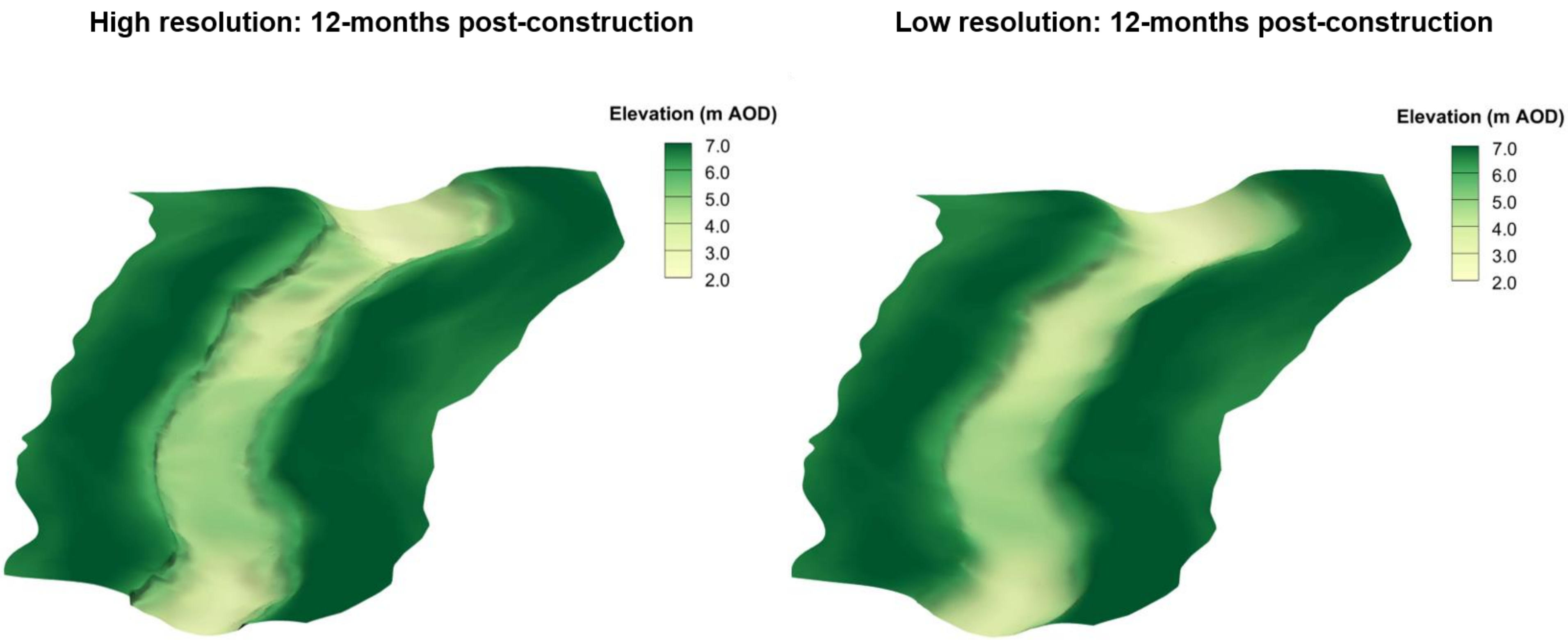


Figure 7.13 Demonstration of 3D visualisations of the 12-month post-construction survey DEMs derived from high and low-resolution data.

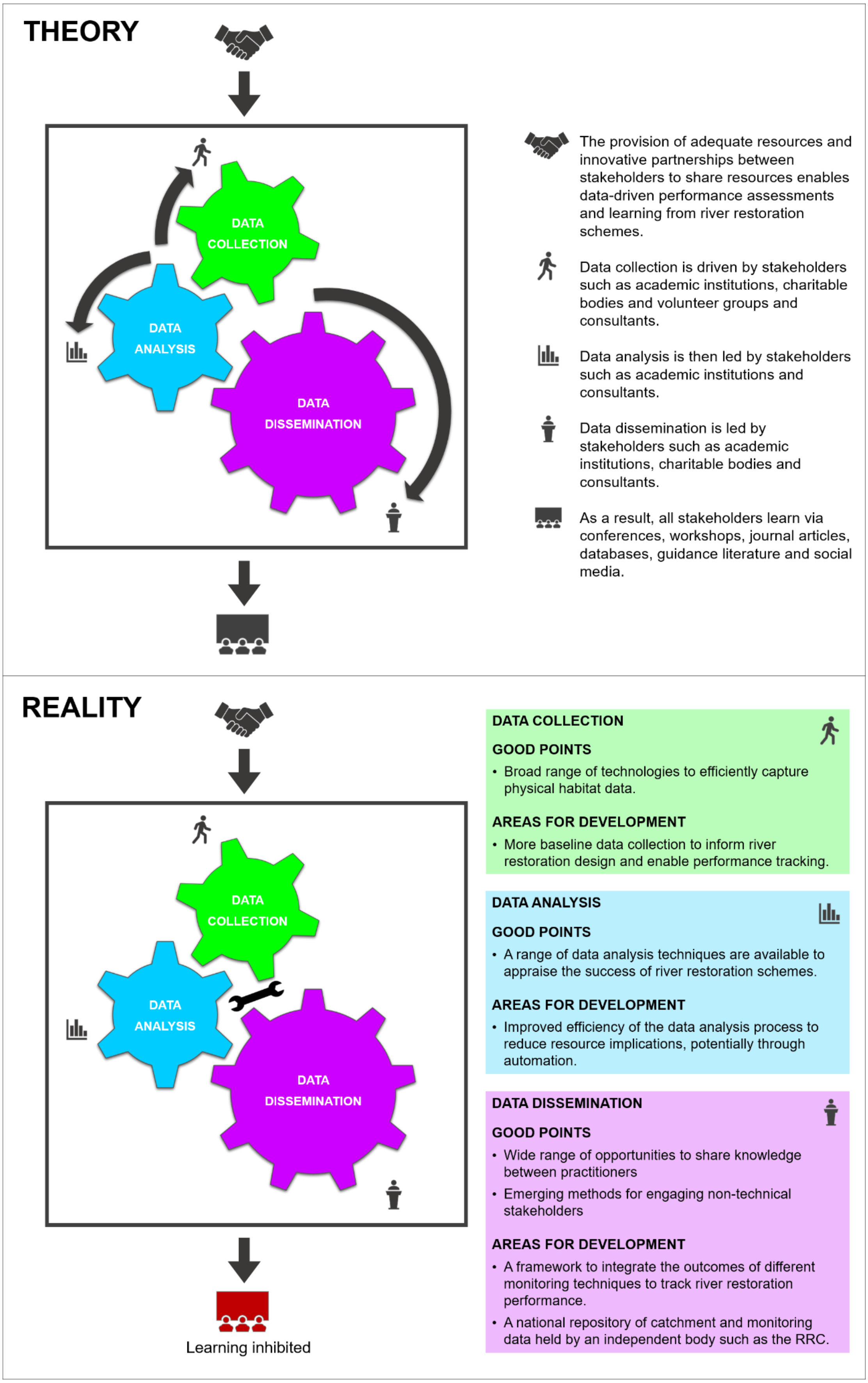


Figure 7.14 The theoretical vs the real state of data-driven performance assessments, a summary of observations from Chapter 2 and Sections 7.2 and 7.3.

7.4 Practical recommendations for data driven performance assessments

This thesis has demonstrated that the ADCP may be a valuable tool for data-driven performance assessments and that higher resolution data sets are valuable learning resources. Data-driven performance assessments should be undertaken for the evaluation of geomorphological and physical habitat data where possible. However, there are some limitations associated with resources when performing these assessments in practice. This section discusses and presents recommendations for two of these limitations, namely; resource constraints and performance tracking.

7.4.1 Recommendations for targeting resources for regulatory and funding bodies

Funding is too often a constraint of river restoration monitoring, particularly for longer-term more resource intensive programmes (Mainstone and Wheeldon, 2016). Consequently, data-driven performance assessments are rarely undertaken outside of academia (Chapter 1 and 2). This research recommended to stakeholders that the monitoring protocol should be extended beyond the initial 18-month monitoring periods, but this was immediately met with a question on which stakeholder would fund it. Therefore, this research supports recommendations that monitoring is funded over the longer-term (Addy et al. 2016) and as part of the initial funding application (Downs & Thorne, 1998).

The value of river restoration monitoring has slowly become realised, and many resource-efficient monitoring methods have been developed to overcome resource constraints (Chapter 2). The uptake in monitoring may have been bolstered by its inclusion as a core component of funding applications. It is recommended here that the value of data-driven performance assessments over less resource intensive methods are highlighted again to the restoration community. Otherwise, there is a risk that low-cost monitoring may become a tick box exercise to satisfy funding requirements. Regulatory and funding bodies could play a larger role in ensuring monitoring is suitable and funds are allocated efficiently, especially as they are well situated to do this during larger funding calls.

England et al. (2008) suggested guidance for the strategic monitoring of river restoration schemes, indicating that resources should be targeted based on the risk of the scheme, and linked to the scale and novelty of the technique. UK best practice guidance also adopts a similar approach (RRC, 2011), but does not fully account for the technological advancements of the last decade which have improved the capacity for monitoring. It also does not recognise that smaller, reach scale projects are typically the norm (Skinner & Bruce-Burgess, 2005; Smith et al., 2014; Castillo et al., 2016) and that river restoration schemes are largely opportunistic (Palmer, 2008). Consequently, these types of schemes may be more representative of river restoration practice but are rarely monitored. However, whilst larger-scale schemes are typically costlier, smaller schemes have been noted to account for a larger proportion of total river restoration expenditure (Bernhardt et al., 2005). Therefore, there is also value in monitoring smaller-scale schemes, particularly as there appears to be limited detailed monitoring of established river restoration design methods. In addition, it may be easier to identify change associated with a single design as part of a smaller scheme, as opposed to when multiple designs are used as parts of a larger scheme.

The RHES, for example, was a small scale and relatively low-cost scheme, it made use of the widely-used principle of gravel augmentation for spawning habitat restoration. However, the design was somewhat experimental in a novel environment. There appear to be few comparative studies to which the performance may be compared (Chapter 6). The exploratory monitoring programme of the RHES has provided valuable learning insights into the sustainability of this technique, design of similar features in the future, adaptive management within the catchment and wider river restoration monitoring practice. Without the monitoring programme detailed within this thesis, fixed point photography would have been used to capture change over time. It is unlikely that these learning outcomes would have been derived from this method as most of the change within the reach was not apparent from above the water surface.

This research has reiterated the importance of baseline monitoring and how data should be collected at the highest resolution practical to inform suitable objectives (Section 7.2). However, it may be difficult to fund high resolution baseline monitoring for all schemes. Typically, restoration projects would be funded subject to the definition of aims and objectives within the funding application. Two potential suggestions for funding bodies are recommended here. First, the development of a fund for exploratory baseline monitoring. This is a potential ‘seed’ fund that could support the development of full applications for restoration funding. Second, funds which already award grants to implement restoration schemes, could potentially release funds for construction on the proviso of an adequate baseline monitoring period and the refinement of suitable objectives and designs. Although, in the latter case, funding streams may often be politically driven and become available at short notice, as well as requiring project completion within a reduced time frame to meet legislative objectives. Therefore, in these situations there is often little time for baseline monitoring. For example, the catchment restoration fund required the completion of the RHES by 2015 to work towards WFD requirements.

Funding is likely to be an issue for undertaking data-driven performance assessments for river restoration schemes until the process becomes more resource efficient or more funding becomes available (Section 7.3). A decision support tool is presented in Figure 7.15 to help identify schemes which present the best 'investment' for post-project data-driven performance assessments. However, if a data-driven performance assessment is a feasible option (e.g. potential to apply for further funding or collaboration with a university) it should be undertaken as it will likely produce the most beneficial outcomes for assessing performance relative to geomorphological and physical habitat objectives.

The decision support tool first screens schemes to understand if a data-driven performance assessment is relevant to the objectives of a scheme. In this thesis, the term river restoration has been used to refer to in-channel physical habitat restoration, but it is a broad practice which includes a range of techniques (including social engagement and community restoration). A data-driven performance assessment is likely to be most beneficial when significant change on the bed or floodplain is possible because of an intervention. Second, the tool screens the novelty of the design components. The tool adopts the approach that establishing a broader knowledge of a range of designs is preferable over in-depth knowledge of just a few designs. While no design will respond in exactly the same way, a sufficient knowledge base established for a type of design should reduce the uncertainty around its response. There is a critical need to review the current knowledge of existing designs to make an informed decision on whether sufficient knowledge has been established.

This research indicated that analyses using low resolution data were still able to detect broad geomorphological and physical habitat trends from a river restoration scheme but the output was not as detailed as from high-resolution data. Consequently, if a scheme requires highly accurate assessments of geomorphological change or physical habitat fragmentation, a data driven performance assessment using an appropriate resolution and technology (Section 7.3.1) is recommended. If multiple schemes are found to be worthy of a data-driven performance assessment, it is recommended here that the priority of resources should be to schemes which have a stronger stakeholder component. This aims to ensure that resources will promote river restoration to the largest possible audience.

Decision support for allocating resources for post-project data-driven performance assessments to maximise the learning potential from river restoration monitoring

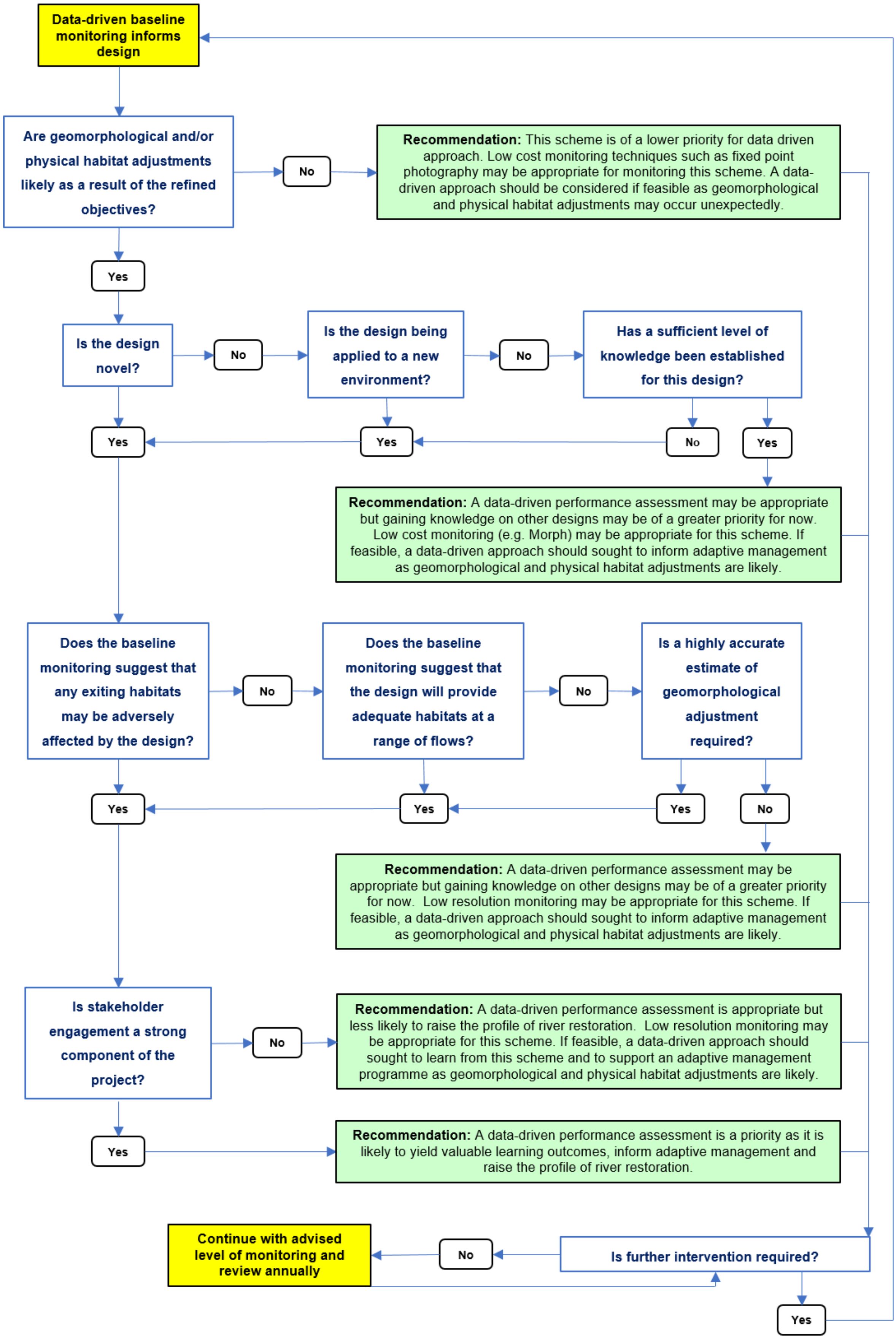


Figure 7.15 Decision support framework for allocating resources for post-project data driven performance assessments

7.4.2 Synthesising data-driven performance assessments

Disseminating the outcomes of river restoration monitoring is critical to learning and adaptive management (Chapter 1). The integration of data-driven monitoring outputs within non-technical methods of dissemination, such as guidance literature, databases and social media may be useful but is not necessarily straight-forward. These methods of dissemination are typically less specialist, more interdisciplinary and case-study focused (Chapter 2). It is not beyond reason to assume that data-driven monitoring may be only one component of a multifaceted monitoring programme. This thesis has referred to river restoration as physical habitat restoration, but river restoration schemes are increasingly interdisciplinary and may have multiple objectives (Chapter 1). Consequently, there may be a need to assess the overall performance of a scheme which has a data-driven component within its monitoring programme. In this situation, the results may need to be reduced, synthesised and integrated with other monitoring data sets.

A lack of appropriate tools to interpret meaningful results may limit participation in river restoration monitoring (England et al., 2008, Chapter 1). Various techniques have been developed for exploring individual objectives. For example, Riverine Community Habitat Assessment and Restoration Concept (RCHARC) which compares velocity and depths to a reference reach (Nestler et al., 1998), Lorenz curves for tracking morphological diversity over time (Soar, 2015) or the HMID (Gostner et al., 2013). A recurring problem is how these tools can be applied repeatedly to define success in different situations. The HMID, for example, was developed in alpine streams and therefore may not be widely applied to other environments (Gosnter et al., 2013). It may be difficult to broadly define success as there are various river types that are restored under varying constraints. Evaluating projects based on technical criteria may not always be appropriate, a full ‘technically’ excellent design may not be achievable nor desirable when balancing the needs of society and conservationists (McDonald et al., 2004; Wharton and Gilvear, 2007; Poff et al., 2016). Thus, SMART objectives (Chapter 2) which are specific to each river restoration scheme, and not a discipline, are critical in assessing performance.

There is need for an overarching framework to draw together multiple indicators to track and communicate the performance of a multidisciplinary scheme. Based on observations made throughout this study, a conceptual performance tracking framework is presented. SMART objectives are a core component of the design of this framework. Whilst it may be difficult to define SMART objectives, there are tools available that could help define objectives of ‘mutually acceptable performance’ for society and conservationists through decision-scaling (Poff et al., 2016). A lack of objective setting has been suggested as a major failing in river restoration monitoring (England et al., 2008). However, with the further promotion of SMART objectives and the concept of a framework that may demonstrate their benefit to the wider community, this monitoring practice may improve.

Assessing performance over longer-time scales is also critical (Roni et al., 2008) as river restoration schemes may have a limited lifespan and require adaptive management. This concept of a performance tracking framework will demonstrate how SMART objectives could be used to assess changing expectations of a scheme over time (i.e. moving targets (Brierley and Friars, 2015)) and help identify if further intervention may be required. The concept should also inform the decision support tree presented in Section 7.4.1 to guide monitoring practice on when further intervention may be required. Importantly, this this section identifies how the framework could integrate with existing learning platforms (e.g. RiverWiki) for non-technical dissemination of river restoration schemes.

If this concept is accepted by members of the wider river restoration community, it may help guide and communicate the findings of river restoration monitoring. This could be developed into an open source graphical user interface (GUI). The framework consists of three stages, the first should be implemented at the outset of a project to guide the setting of SMART objectives which are critical to defining success (Fig. 7.16). A period of baseline monitoring should identify river restoration goals and objectives over short (< 5 years), medium (5-10 years) and long (> 10 years) time periods. Each objective should have at least one suitable and specific indicator that it can be measured against. Two values should be set for each indicator. The first value should be set at a level indicating project success ('success'). The second value should be set such that if the indicator fails to reach this level, further intervention should be considered ('further intervention'). A result falling between the two values indicates that monitoring of this indicator should continue, but immediate intervention is not yet required ('further monitoring').

This framework recognises that some river restoration objectives may be more important to the overall aim of the project and that this importance may change overtime, or objectives may account for a greater proportion of project expenditure. Therefore, at this stage, the framework allows the user to weight objectives according to their preference. The baseline data sets and SMART objectives would be recorded within the GUI. Following the setting of SMART objectives, the post-project surveying and analysis is undertaken following guidance (Section 7.4.1 and 7.3.1). Once the first set of post-project monitoring has been undertaken, the second phase requires the user to enter the data into the GUI, and they are automatically plotted on to a series of graphs (Figs. 7.17 and 7.18). These results are evaluated against the SMART objective performance criteria.

The performance is broken down as either success, further monitoring, further intervention or no data, and displayed on a pie chart with the size of the slice representing the weighting of the objective. The overall assessment of performance may be categorised as successful if all the monitoring results within a specified timeframe meet or exceed the SMART objective criteria. When some criteria are not met, the overall performance is determined by the monitoring result of the lowest category.

These visuals were conceptualised such that they were easily interpreted by the non-expert and may be integrated with existing learning platforms in the third stage of the framework (Fig. 7.19). The overall pie charts could be shared on social media and provide links to online databases where information on project performance is available in more detail and where data sets (and other information) are stored. By clicking on a slice of the chart, the user could be directed to the specific monitoring data used to derive the overall performance of that specific objective. This concept is presented here for consideration by the river restoration community for discussion, and as such a SWOT analysis of this technique is provided in Table 7.6. The concept is demonstrated in the next section.

Table 7.5 Strengths Weaknesses Opportunities and Threats of the performance tracking framework concept

Strengths	Weaknesses
• Performance is evaluated relative to objectives rather than technical criteria. • Potential to integrate interdisciplinary monitoring to identify project success. • Performance may support adaptive management by identifying the need for further intervention.	• Does not provide an opportunity for disseminating lessons learnt unexpectedly. • Difficult to distinguish by non-experts if project success or failure is due to the level of rigor of the SMART objectives.
Opportunities	**Threats**
• The framework could integrate with existing learning platforms (e.g. RiverWiki and Social Media) to help non-technical stakeholders interpret project performance. • The framework may raise awareness of setting SMART objectives of river restoration projects.	• Users setting easily achievable objectives – need to praise and reward learning of both success and failures. • Adequate resources to support baseline monitoring for informing SMART objective setting and to support long-term monitoring.

Performance Tracking Framework: Stage 1 – Defining Objectives

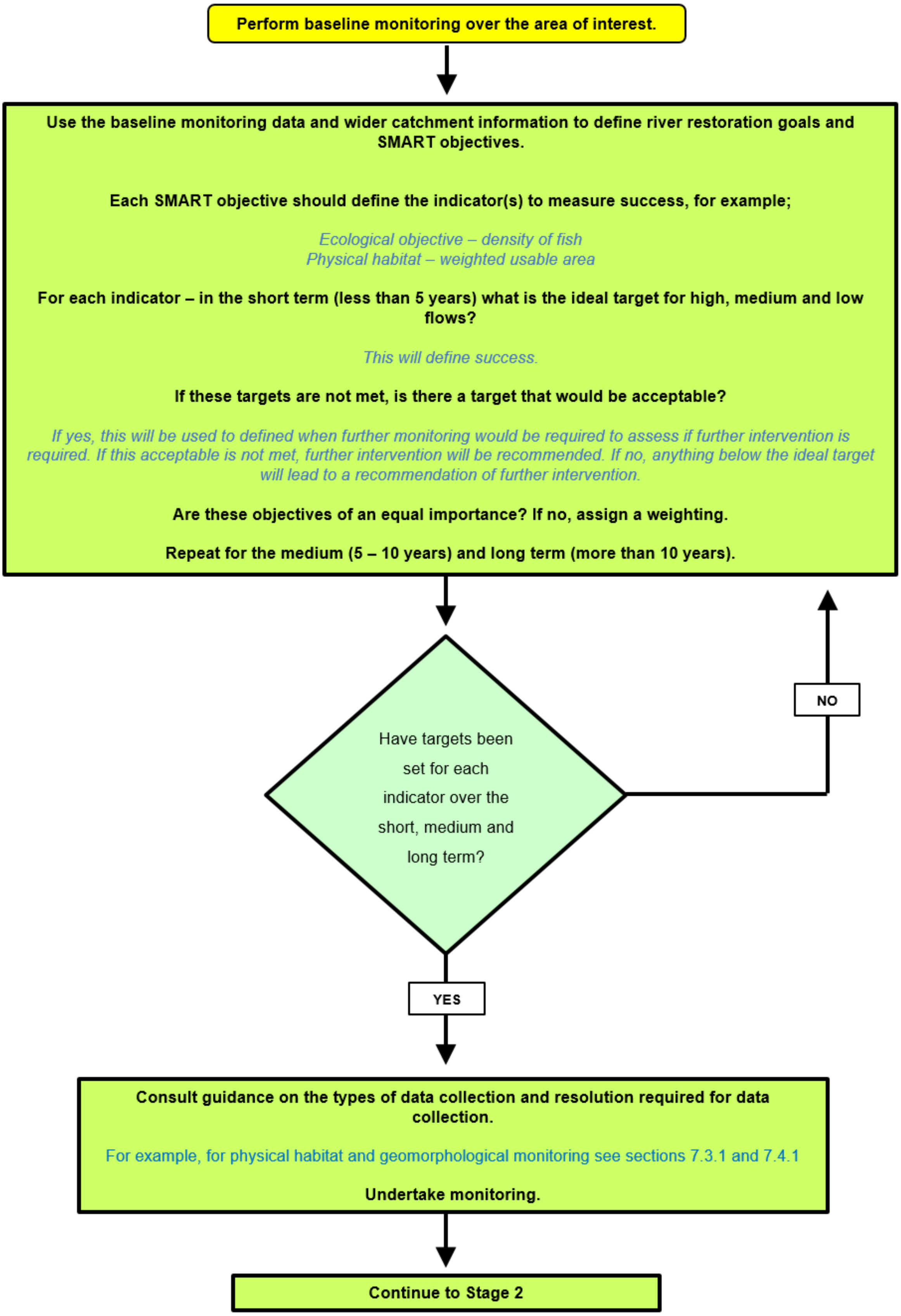


Figure 7.16 Stage 1 of the river restoration performance tracking framework concept – defining objectives.

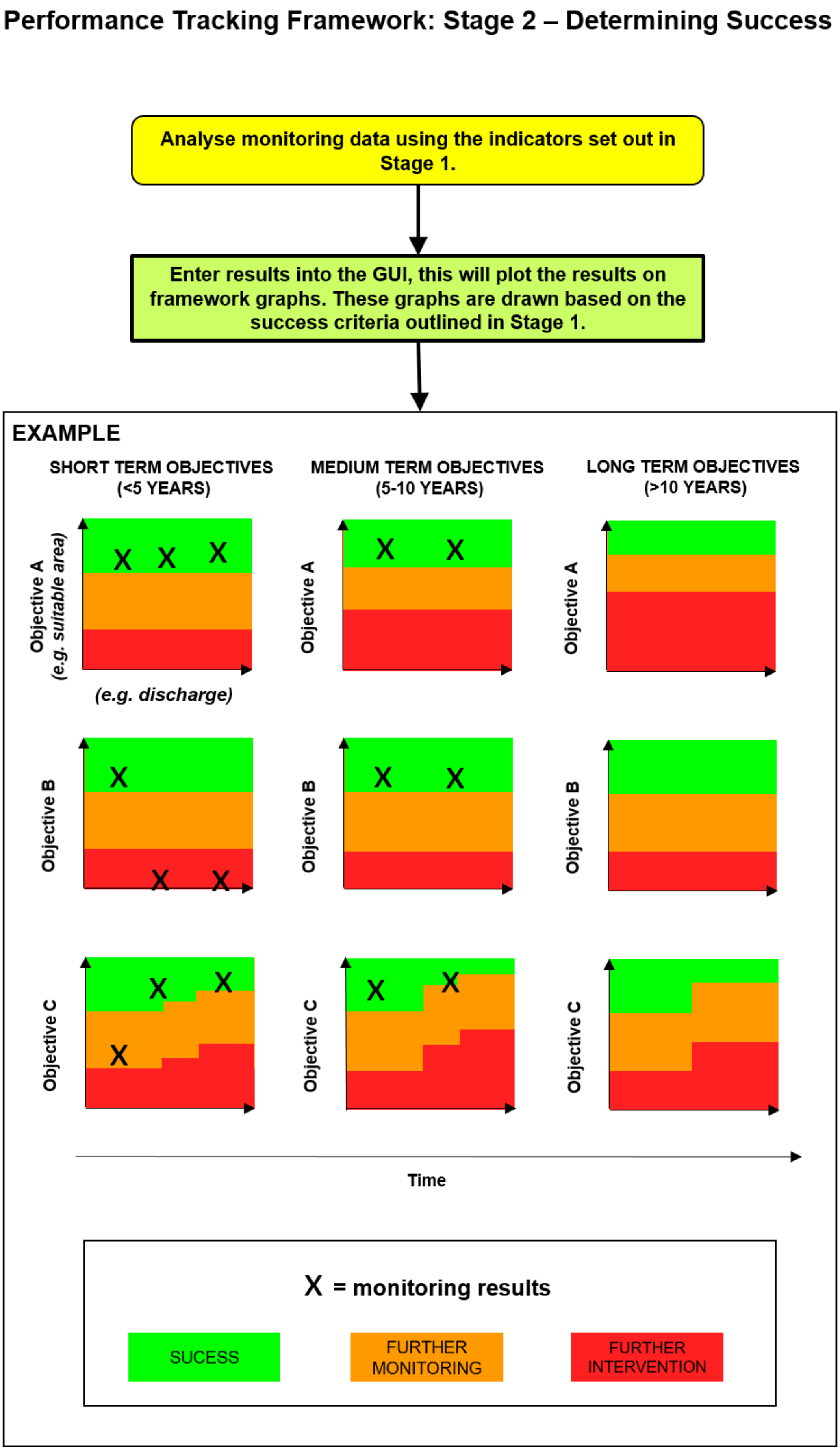


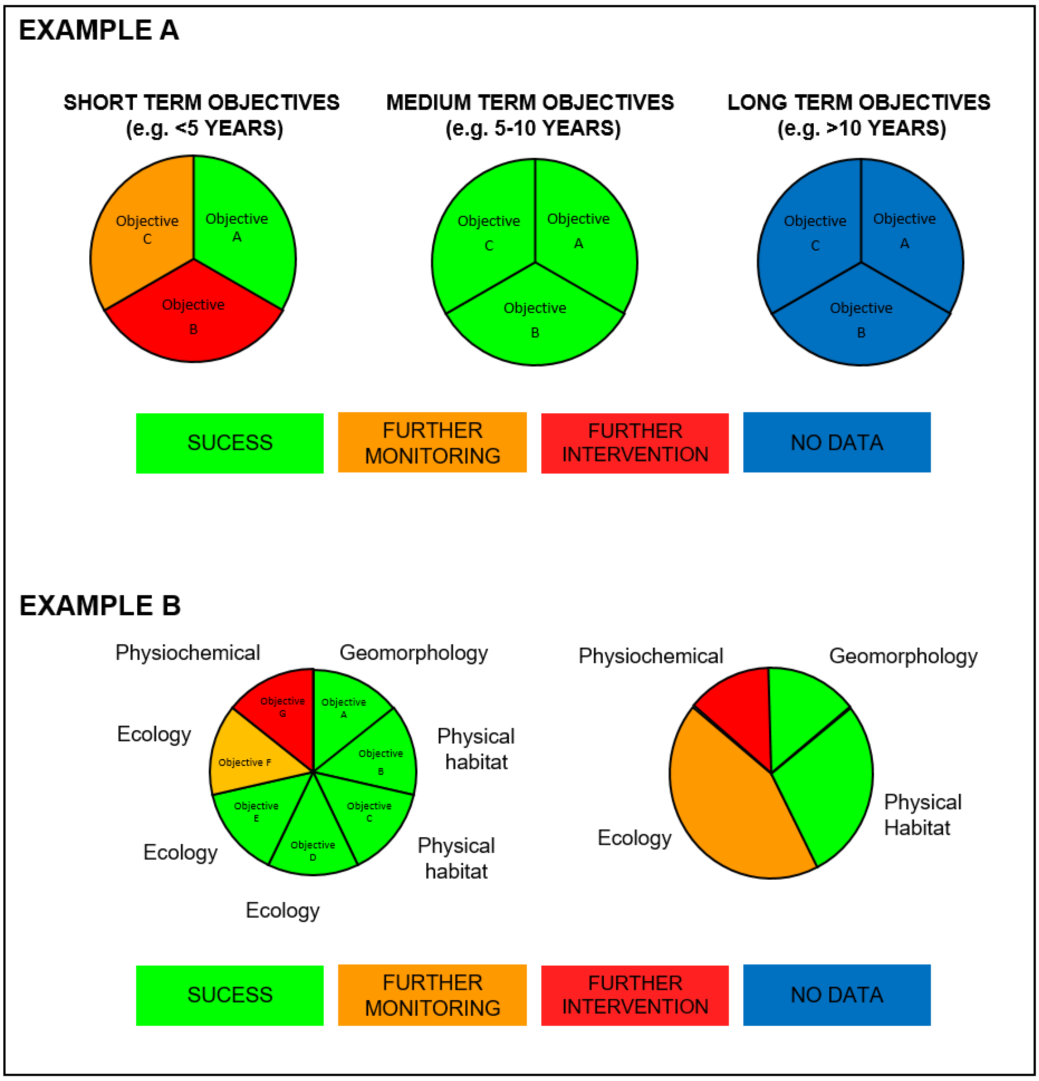
Figure 7.17 Stage 2 of the river restoration performance tracking framework concept – determining success.

The framework will then decide, based on the SMART objective criteria, if the results suggest;

SUCESS | FURTHER MONITORING | FURTHER INTERVENTION | NO DATA

All monitoring results within a specified timeframe must be categorised as successful for an overall 'successful performance'. When not all of these results are successful, the overall performance is determined by the monitoring result of the lowest category.

The results are then plotted within a pie chart so that the overall performance may be easily interpreted (Example A). If multiple objectives are used, these may be grouped into discipline (e.g. ecology, physical habitat, geomorphology) and categorised in a similar method (Example B).



Continue to Stage 3

Figure 7.18 Stage 2 of the river restoration performance tracking framework concept – determining success.

Performance Tracking Framework: Stage 3 – Dissemination

Disseminate the performance of the scheme widely, an example is given below to demonstrate how this could be approached and the framework could be integrated into existing learning platforms.

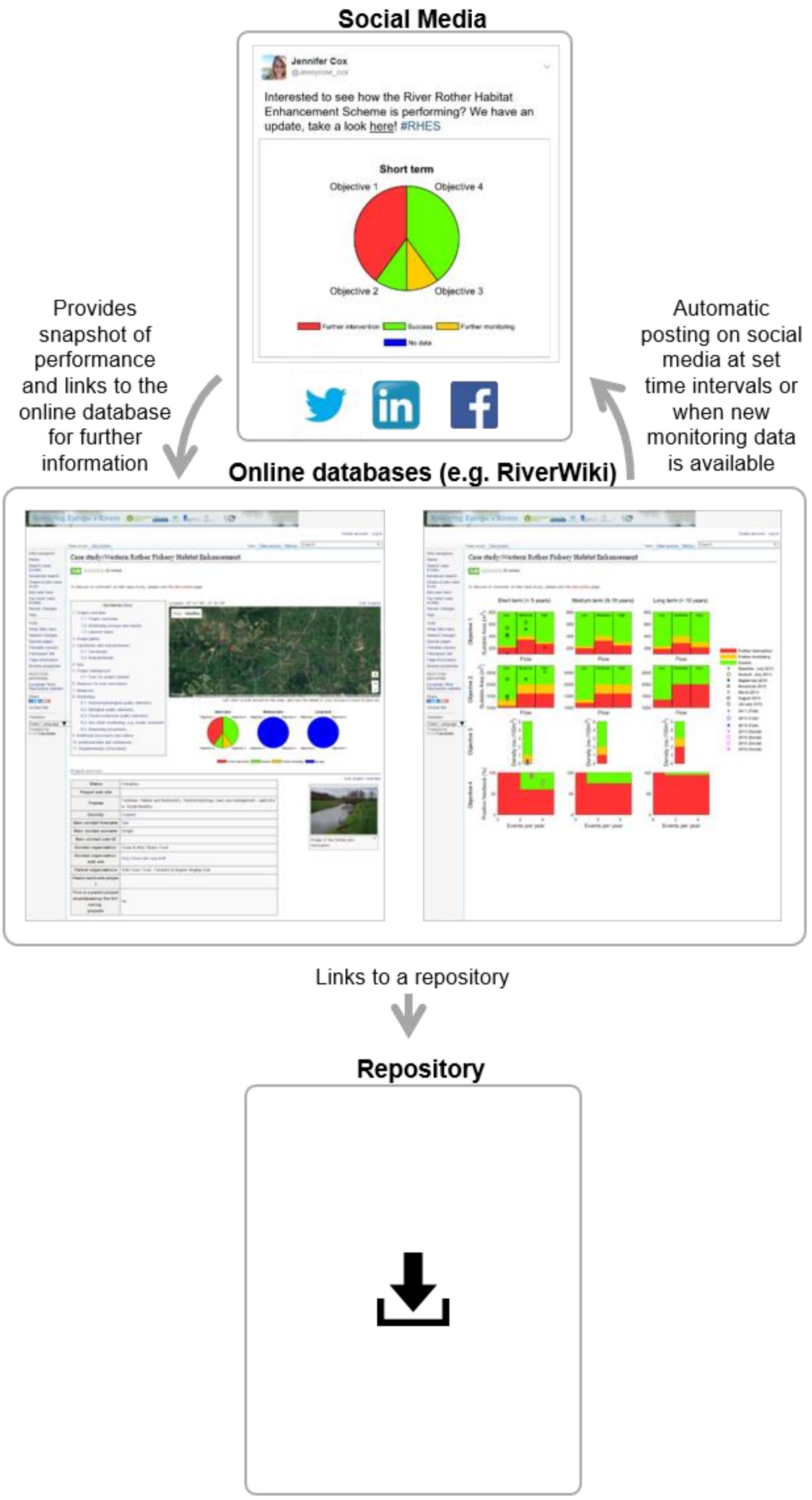


Figure 7.19 Stage 3 of the river restoration performance tracking framework concept.

Demonstration of the performance tracking framework concept

The RHES data are used in two simple scenarios to demonstrate this concept. Suitable SMART objectives for this demonstration of RHES were not outlined at the beginning of the project. Therefore, these scenarios are ***fictional***, intended only for demonstration purposes and thus ***should not*** be used to judge the performance of the RHES. The first, demonstrates the physical habitat and geomorphological performance of a scheme over the short term (< 5 years). The second, presented in Appendix E, demonstrates how objectives may change over time and how data from a range of river restoration objectives (e.g. physical habitat, ecological and social) may be integrated. Further data collection and testing of the concept is required but beyond the scope of this research study.

Objectives for the first scenario are presented in Table 7.7. If applied to future projects, baseline data and design modelling would inform objective setting. This fictional scheme aims to improve the geomorphological diversity of the target reach and improve physical habitat for brown trout. The reach afforded adequate physical habitat for juvenile brown trout prior to restoration, and the scheme aimed to maintain this. The remaining life stages of brown trout were not adequately provisioned and thus, the aim was to improve the abundance for these life stages. Physical habitat suitable for spawning and fry brown trout were least abundant within the catchment, and thus received a greater weighting. The reach was geomorphologically uniform prior to restoration and a riffle feature was constructed to improve diversity and meet the physical habitat objectives of this scheme.

The results (Figs 7.20 & 7.21) indicate that overall the geomorphological objectives were met and the design was successful with respect to these criteria. The results suggest a mixed performance relative to the physical habitat objectives. Adequate spawning habitat was not found in sufficient quantities to allow this objective to be determined successful. However, it was found to be in the band of uncertainty defined in the objectives, and consequently further monitoring is recommended to establish if further intervention is required. The abundance of physical habitat suitable for fry, however, was found to be consistently inadequate and further intervention is recommended. Conversely, the remaining objectives were adequately provisioned over the short term and the design was identified as successful relative to these objectives. This example demonstrates that this framework could identify and justify adaptive management within river restoration. If adaptive management is required, high resolution monitoring would be recommended (if practical and not performed as part of the monitoring programme to date) to inform adequate design criteria.

Restoration type	Objective	Indicator	Baseline	Further monitoring (minimum)	Success (minimum)	Time frame	Weighting
Geomorphology							**40%**
Elevation	Improve diversity	Standard deviation (reach scale)	0.45 (low flow) 0.45 (mod flow) 0.45 (high flow)	0.50 (low flow) 0.50 (mod flow) 0.50 (high flow)	0.53 (low flow) 0.53 (mod flow) 0.53 (high flow)	Short-term (< 5 years)	20%
Velocity	Improve diversity	Coefficient of variation (reach scale)	0.66 (low flow) 0.55 (mod flow) 0.50 (high flow)	0.70 (low flow) 0.60 (mod flow) 0.55 (high flow)	0.82 (low flow) 0.68 (mod flow) 0.60 (high flow)	Short-term (< 5 years)	20%
Physical habitat							**60%**
Spawning brown trout	Improve physical habitat abundance	Suitable area (m^2) based on preference criteria (reach scale)	115 (low flow) 200 (mod flow) 150 (high flow)	300 (low flow) 250 (mod flow) 200 (high flow)	350 (low flow) 300 (mod flow) 250 (high flow)	Short-term (< 5 years)	25%
Fry brown trout	Improve physical habitat abundance	Suitable area (m^2) based on preference criteria (reach scale)	58 (low flow) 50 (mod flow) 50 (high flow)	200 (low flow) 200 (mod flow) 200 (high flow)	300 (low flow) 300 (mod flow) 300 (high flow)	Short-term (< 5 years)	15%
Adult brown trout	Improve physical habitat abundance	Suitable area (m^2) based on preference criteria (reach scale)	318 (low flow) 1000 (mod flow) 1000 (high flow)	340 (low flow) 1100 (mod flow) 1100 (high flow)	390 (low flow) 1200 (mod flow) 1200 (high flow)	Short-term (< 5 years)	10%
Juvenile brown trout	Maintain physical habitat abundance	Suitable area (m^2) based on preference criteria (reach scale)	925 (low flow) 1000 (mod flow) 1000 (high flow)	925 (low flow) 1000 (mod flow) 1000 (high flow)	925 (low flow) 1000 (mod flow) 1000 (high flow)	Short-term (< 5 years)	10%

Table 7.6 SMART objectives for the demonstration of the performance tracking framework

Results:

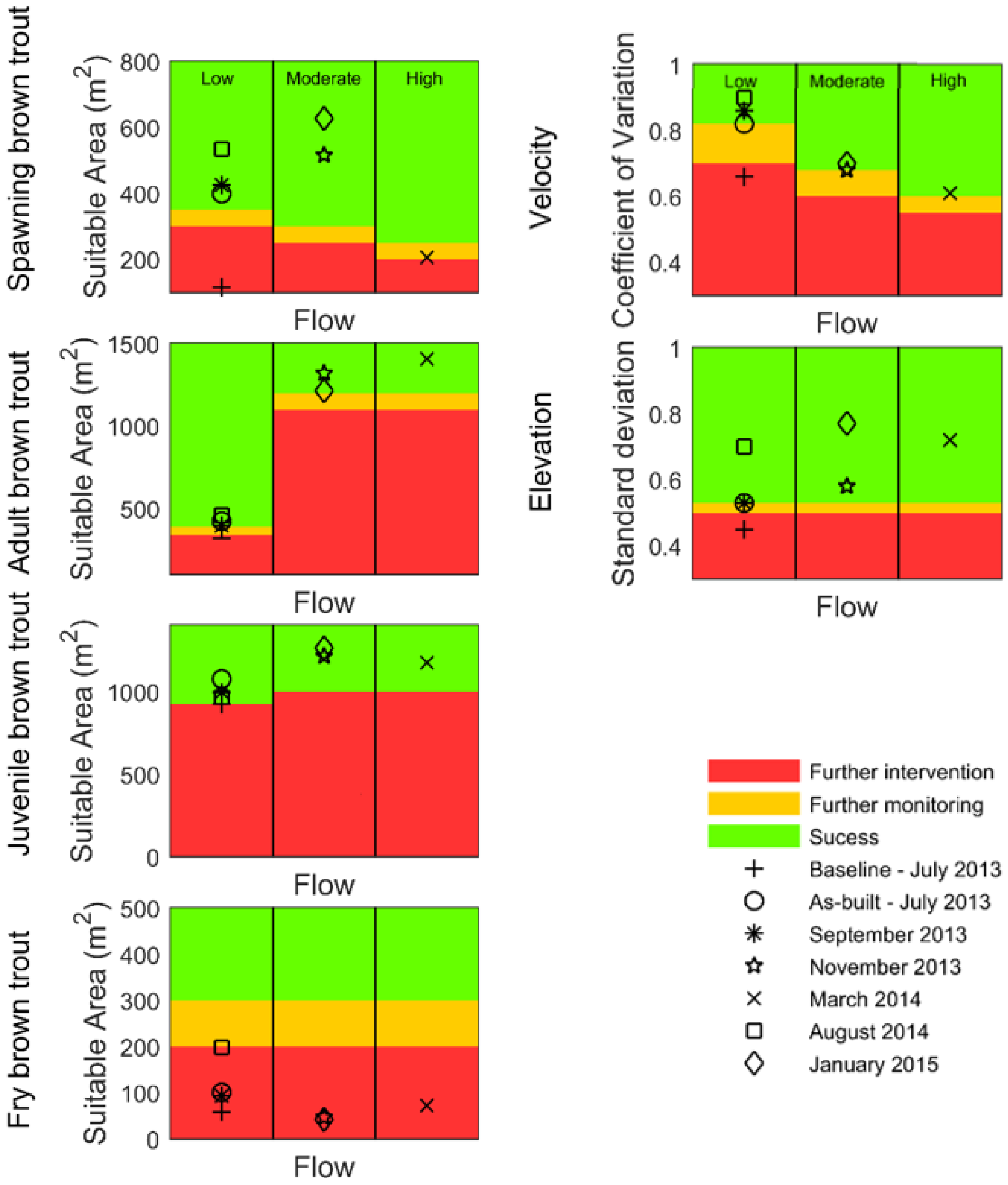


Figure 7.20 Possible output of the framework. Results of the river restoration scenario presented to demonstrate the performance tracking framework.

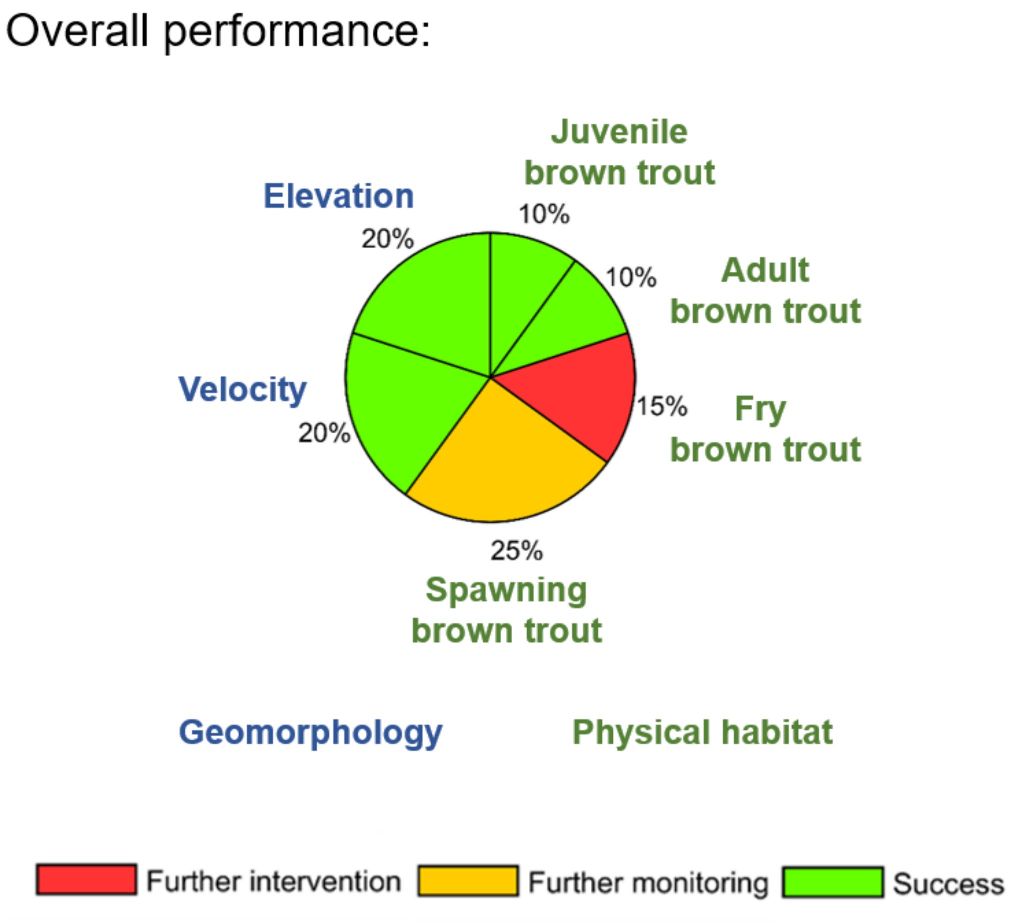


Figure 7.21 Possible output of the framework. Overall performance of the river restoration scenario presented to demonstrate the performance tracking framework.

7.5 Key findings of Chapter 7

This chapter has explored the value of high-resolution river restoration monitoring data. The evaluation of high and low-resolution data indicates that low-resolution data may provide less accurate representations of geomorphological and physical habitat forms and changes. High-resolution spatial data may be particularly useful in detecting fragmentation of physical habitats as well as providing generally more accurate results. Additionally, high-resolution vertical data provided the opportunity to assess the effect of different velocity representations on the results of the physical habitat analysis. These results varied significantly and there is a need to improve our ecological understanding of velocity suitability within the water column, to refine physical habitat assessments.

Overall, if possible, high-resolution data collection is recommended for all river restoration monitoring, particularly during the pre-project baseline monitoring period. However, it is acknowledged that will not always be possible due to resource constraints. This should help facilitate the identification of suitable design objectives and maximise a scheme's performance. At present, there appears to be limited scope to disseminate data sets widely beyond peer-reviewed journal articles and conferences. This is a concern as these techniques become less novel, the scope for disseminating within these spheres may become harder. The need to share data, potentially using existing platforms (e.g. the RiverWiki), has been highlighted by the potential need to re-assess data if schemes that have been monitored at different resolutions are to be compared.

Low resolution data may still be beneficial for detecting broad geomorphological physical habitat trends of river restoration schemes, that may be missed through fixed point photography. The ADCP has the potential to improve the efficiency of data collection of physical habitat variables at both high and low resolution. Consequently, the ADCP may improve the ability to capture physical habitat variables in environments in which change may occur rapidly (e.g. catchments with a flashy flood hydrograph). The chapter presented guidance for data collection using the ADCP, UAVs and traditional methods for river restoration monitoring (Section 7.3.1). UAVs are likely to be the most-resource efficient method for river restoration monitoring, but are limited in their application. ADCPs may be a suitable alternative as suggested by Marteau et al. (2016) and as demonstrated within this thesis, but their application is also limited in certain situations. Traditional point measurements may still be required in some environments.

It is highlighted here that whilst the resource efficiency of capturing data for data-driven performance assessments may be improving, the post-processing and analysis of these data may have significant resource implications. Without further development of software to support these processes, data-driven performance assessments are likely to remain the pursuit of research institutions. It is realistic that in practice data-driven performance assessments are likely to be restricted by resource constraints. Guidance, targeted primarily at funding bodies, is presented to facilitate resource allocation and maximise the learning opportunities from river restoration schemes (Section 7.4.1). This research has highlighted that data-driven performance assessments may be particularly useful for developing non-technical engagement tools (Section 7.3.3). Therefore, where resources are limited and there are multiple projects that may require high-resolution post-project monitoring, it is recommended that resources are targeted at projects with a strong stakeholder engagement component.

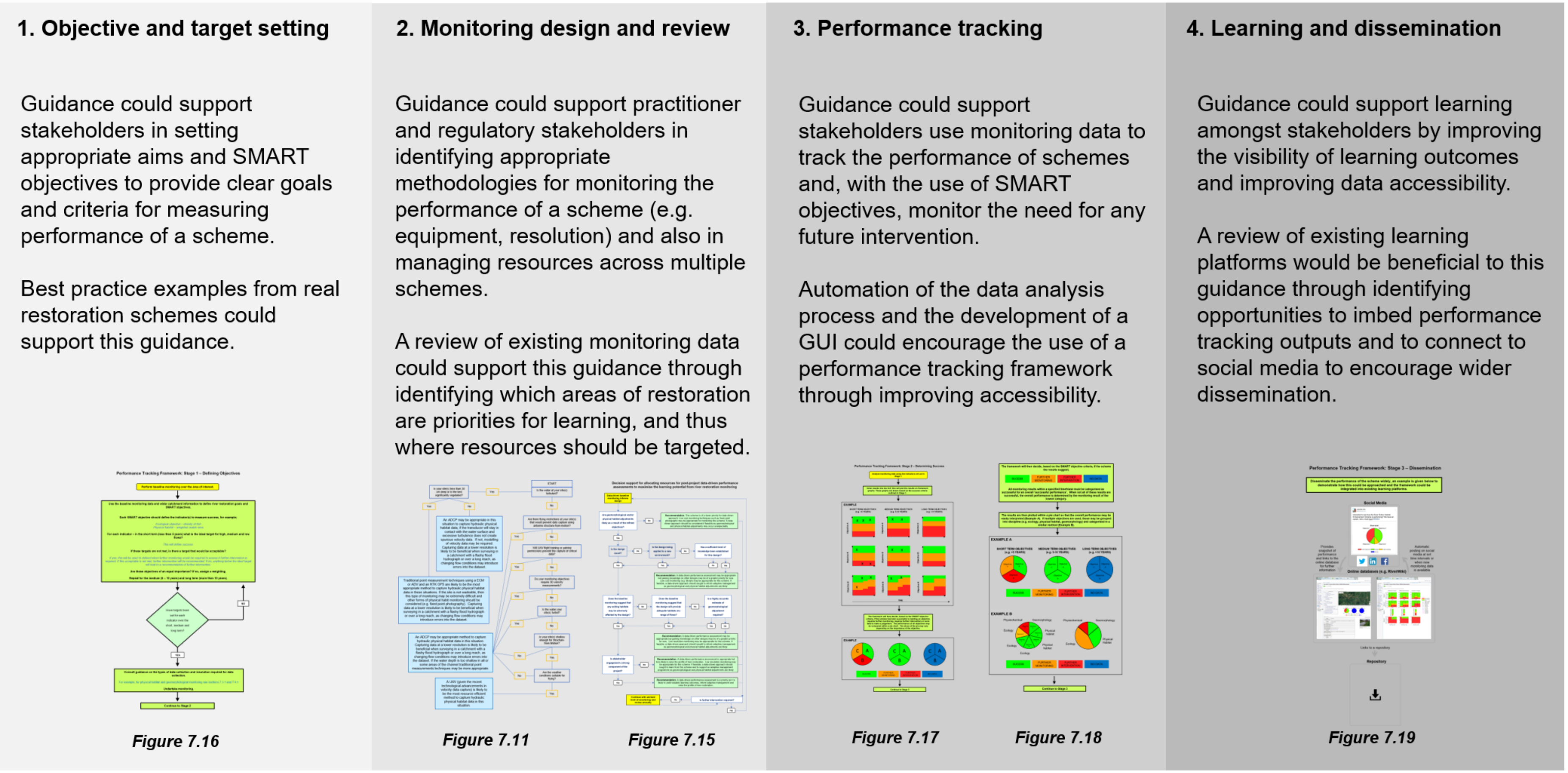


Figure 7.22 Summary of guidance presented in Chapter 7 that could support the river restoration monitoring process, and a summary of opportunities to further develop this guidance.

River restoration schemes are typically highly interdisciplinary and have multiple objectives. Performance tracking tools to facilitate the interpretation of overall project performance by non-experts are needed. A concept of a framework to facilitate this is presented to integrate geomorphological and physical habitat data-driven performance assessments with the monitoring results of other river restoration objectives. This framework would require further development but presents opportunities to guide adaptive management and integrate with existing learning platforms to disseminate project performance to a wide range of stakeholders. This framework does not facilitate disseminating lessons learnt by 'surprise', which is a limitation. In addition, SMART objectives are critical to this framework and there is a risk that schemes could set easy objectives to ensure a positive project performance.

The suggestions made within this chapter are not instructional to the wider river restoration community but are presented for discussion. The guidance presented here could be used to support the river restoration monitoring process. The practical implementation and value of this guidance is summarised in Figure 7.22. Ideally, the individual aspects of the guidance would be developed further into an interactive process with a GUI, which would encourage standardisation between monitoring programmes and improve usability of the guidance. The process starts with objective and target setting to develop performance criteria that are suitable for the environment and stakeholder requirements. Continuing from this initial step, the process would support the design and review of the monitoring programme to ensure the selection of appropriate technology given the resources available. Following an appropriate period of monitoring, data collected during this process can be used to track the performance of the scheme and monitor the need for any further intervention. Ultimately, the process would provide outputs for effective learning and dissemination of performance amongst stakeholders by linking results to online databases and social media.

8 Conclusion

8.1 Research Overview

This thesis aimed to improve the potential for learning and adaptive management within river restoration practice. More specifically, this research aimed to explore and present guidance for how cost-effective data collection, processing, analysis and communication of geomorphological and physical habitat datasets may be more routinely applied within river restoration. To achieve the aim of this research, five specific objectives were outlined and met within Chapters 2-7. The first objective was to '*evaluate existing methods of data collection and analysis of physical habitat variables used in related monitoring studies to identify novel technologies for river restoration monitoring'.*

To meet this objective, a review of existing data collection methods for geomorphological and physical habitat monitoring was undertaken and reported in Chapter 2. This review suggested that emerging technologies are widely and routinely used in academia to advance the scientific knowledge of river environments. However, the application of these technologies in river restoration monitoring outside of academia is rare. Some low-cost methods of capturing spatially continuous elevation data (e.g. Structure-from Motion) are limited by operational capabilities in some river environments. As an alternative, the Acoustic Doppler Current Profiler (ADCP) had been suggested as a suitable method for river restoration monitoring, but there was a need to critically evaluate the practical use of this technology in this context. Consequently, the ADCP was identified as a novel technology for river restoration monitoring that was to be evaluated within this research.

The review of data analysis methods indicated that there were a range of methods available to evaluate geomorphological and physical habitat data. However, the guidance on how to undertake these more technical analyses was limited. Some academic studies publish code alongside their work, but this has not been the norm and still requires technical ability to utilise this published material. Most of the studies documented within the literature appear to evaluate modelled physical habitat data, rather than real data. This may reflect the difficulty in collecting real data. The simulation of physical habitat availability using this modelled data was a popular technique documented within the literature. However, these analyses mostly focused on evaluating changes to the abundance of physical habitat and rarely sought to evaluate the quality of physical habitat over time. Therefore, the novel analysis of the quality and fragmentation of physical habitat simulations (using suitability curves) was also identified as an area for exploration within this thesis.

Once the ADCP was identified as a novel technology and the possibility of a potential novel analysis of simulated physical habitat data was acknowledged, the thesis then sought to address the second objective; *"to explore the practical use of a novel technology for collecting physical habitat variables in the context of a case-study river restoration scheme",* and the third objective; *"to assess the geomorphological and physical habitat performance of the case-study restoration scheme over space and time to learn from the scheme, as well as explore the practicality of analysing larger river restoration datasets".* In order, to achieve these objectives a case-study river restoration scheme, the River Rother Habitat Enhancement Scheme (RHES) was identified and evaluated.

In the review of river restoration monitoring practices (Chapter 2), it was highlighted that understanding the historical and present catchment conditions of a river restoration scheme was important for its effective evaluation (Downs and Kondolf, 2002; Downs et al., 2011; Friberg et al., 2016). The author explored the data available for the River Rother Catchment, West Sussex. It was found that an updated analysis and synthesis of this data would be beneficial for evaluating the geomorphological and physical habitat performance of this project, and for informing wider catchment management. Consequently, Chapter 3 outlined the RHES and the catchment context of the River Rother, so that the scheme could be appropriately evaluated with consideration to the wider catchment processes later within the thesis (Chapters 5 and 6).

An updated analysis of the data relevant to the River Rother catchment suggested that potentially too much emphasis had been placed on catchment wide arable agricultural sources of fine sediment. It was likely that in-channel geomorphological processes that were partly controlling fine sediment accumulation had been underestimated. It is undoubtable that the erosion of friable soils and the delivery of fine sediment to the channel is a significant factor contributing to the degraded ecological state of the catchment. However, the substantial modification of in-channel catchment processes over the last millennia is of at least equal significance to the current ecological state of the River Rother. An analysis of stream power using Environment Agency (EA) cross-sections suggests that fine sediment is likely to naturally accumulate within the lower catchment of the River Rother and that the channel is probably overwide. The data synthesised and analysed within Chapter 3 does indicate that gravel augmentation proposed by the RHES was likely to be the only feasible method to create suitable spawning habitat for brown trout. Chapter 3 challenges the perception of sediment issues within the case-study catchment and the updated catchment baseline data was critical for understanding (and evaluating) the geomorphological and physical habitat performance of the RHES (Objective 3) in Chapters 5 and 6.

Where feasible, best-practice guidance was adopted in the development an exploratory 18-month monitoring programme of the RHES using an ADCP. The methods of data collection, post-processing and analysis used within this thesis are documented and summarised with a workflow in Chapter 4. This may serve as guidance for future monitoring schemes. Some limitations of the monitoring programme were identified, such as a short period of baseline monitoring and a lack of control site. These limitations may have been addressed if a longer period of pre-project monitoring was available. Consequently, the need for improved forward planning for river restoration was highlighted. Data collection using this technology was straight forward, but some significant post-processing of the ADCP data was required. This process was documented and summarised within the presented workflow.

The results of the geomorphological performance of the RHES were reported and discussed in Chapter 5. The results highlighted that gravel augmentation, a riffle feature implemented as part of the RHES, improved the morphological and hydraulic variability of the full study reach without compromising existing morphological processes. However, the scale at which analyses are undertaken (e.g. feature or reach scale) was found to have the potential to influence the results of the analyses. This is an important consideration for future river restoration monitoring schemes. The succession of flows experienced during the monitoring period provided an opportunity to test the resilience of the riffle feature to a large magnitude event (the 2013/14 floods). The form of the feature was reasonably resilient in flows up to bankfull, but the results suggested that it may not have been fully self-cleansing. This is a potential result of the initial siting of the crest over the tail of the riffle, which may have promoted sediment deposition over the feature. The riffle feature was reasonably resilient to a series of large out of bank flows, but it did experience some structural modification. Following the 2013/14 floods, the riffle feature appeared to be re-shaped and functioning more akin to a natural riffle.

A velocity convergence but not reversal was observed between the constructed riffle feature and the adjacent pools as discharge increased up to bankfull (pre-and post-flooding). The out-of-bank flows experienced within the reach resulted in significant geomorphological change. These flows were not captured due to health and safety risks but data of these flows would have been beneficial to develop a better understanding of pool-riffle maintenance mechanisms. Consequently, the utilisation of remotely operated technologies to capture data at these flows is advocated here. It is not clear from the results if improvements in morphological diversity within the reach were solely down to the riffle placement or in combination with the 2013/14 flood events. This highlighted that a control site would have been beneficial to reduce the uncertainty around these geomorphological processes. This was a limitation identified in Chapter 4 due to a short lead in time to the project.

The feature may have improved the geomorphological performance within the study reach, but there was limited evidence to suggest that it had a significant impact beyond the reach. This was not a failing of the riffle feature as it was designed as a reach scale restoration effort. However, many of the processes affecting the geomorphological performance were operating at the catchment scale and the ability of the riffle feature to be robust to catchment-scale processes over a longer time-period without further intervention was uncertain. A range of recommendations to improve the geomorphological performance of the River Rother were outlined, these included:

- The exploration of catchment scale approaches to manage the in-stream fine sediment load, which may include fencing to reduce bank erosion, sediment traps, riparian buffers and the restoration of floodplain connectivity;
- The restoration of longitudinal sediment connectivity and the consideration of frequent gravel augmentation to sustain riffle features within the lower catchment, if desired;
- The consideration of trialling a riffle feature with the crest sited over the head of the riffle feature, as this is more likely to be self-cleansing, or (if feasible) a range of designs with the view of optimising restoration performance, and;
- The consideration of the installation of a deflector or similar constriction mechanism just upstream of the reach to try to maintain the newly gained morphological diversity within the reach.

The research and recommendations presented in Chapter 5 demonstrate that the ADCP may be used to learn from the geomorphological performance of river restoration schemes (Objectives 2 and 3). Therefore, the ADCP may be used as a suitable alternative if the use of more efficient methods of data collection (e.g. UAVs) may be impractical or infeasible as suggested by Marteau et al. (2016). Nonetheless, adequate baseline monitoring and catchment information are critical to the success of data driven performance assessments, as well as identifying suitable geomorphological river restoration objectives and designs.

The results of the physical habitat analyses of the RHES were reported and discussed in Chapter 6. The results highlighted that gravel augmentation as part of the RHES improved the physical habitat heterogeneity of the reach. However, it was noted that the scale at which physical habitat heterogeneity assessments are undertaken may have a significant impact on the results. For example, physical habitat heterogeneity improved following restoration when evaluated at the reach scale, but physical habitat appeared to become more homogenous when assessed at the riffle scale. This again highlighted the importance of considering the scale of data-driven performance assessments within river restoration.

The monitoring campaign indicated that the RHES was successful in meeting its objective of improving spawning habitat provision for brown trout. However, the potential accumulation of fine sediment within the riffle feature may have limited the features viability as a spawning habitat. A lack of suitable physical habitat for fry was significant both within the reach and suspected within the wider catchment, both prior to and following restoration. The reach afforded abundant physical habitat for juvenile brown trout, but migratory barriers may prevent this physical habitat from being fully utilised. Consequently, whilst the RHES was successful in meeting its objective, substantial further intervention is likely to be required for the River Rother to be able to support an abundant and healthy population of brown trout. Using both the ADCP data in conjunction with the catchment baseline data information (Chapter 3), this chapter posed the question of whether the restoration of the River Rother for brown trout was a sustainable objective for the future. It was suggested that restoration of geomorphological processes and physical habitat in the lower catchment for other coarse fish species may be more sustainable.

The effect of restoration activities on the physical habitat of two species known to be present within the catchment prior to restoration was also evaluated. These species were dace and roach and the results indicated that the physical habitat for both species deteriorated following restoration, particularly for the adult and juvenile life stages. However, given the morphology of the reach is similar to that of the wider lower catchment, it is unlikely that this small decline in physical habitat provision will significantly affect these species. Nonetheless, this research highlights the importance of undertaking high-resolution baseline monitoring to understand and mitigate the potential loss of habitat for non-target species. Future ecological monitoring has been recommended to check if there are any other unintended adverse effects of this restoration. This chapter demonstrated that different methods of physical habitat assessment may afford different interpretations of performance. Therefore, using a suite of analytical methods may be useful for providing an informed assessment of river restoration performance. Gravel augmentation schemes are common within river restoration, although there appeared to be few studies detailing their performance for a comparative analysis. This study encourages the evaluation of similar schemes to improve on best practice within river management.

The research and recommendations presented in Chapter 6 demonstrated that the ADCP may be used to learn from the physical habitat performance of river restoration schemes (Objectives 2 and 3). However, it is acknowledged that research undertaken in an academic institution is not always feasible in industry due to resource constraints. Chapter 7 addressed the fourth objective of this research, this was "*based on the experiences of undertaking a river restoration monitoring programme, to critically evaluate the practical application of routine data-driven river restoration monitoring".* This chapter identified that through resampling the high-resolution geomorphological and physical habitat monitoring data of the RHES, high-resolution data facilitated a more detailed and accurate interpretation of river restoration performance. High-resolution monitoring data was particularly useful in identifying the fragmentation of physical habitats, a component which was found to have a significant effect on the interpretation of physical habitat performance.

However, the interpolation of low-resolution data was still sufficient to detect the broad geomorphological and physical habitat changes that occurred within the RHES monitoring programme. Therefore, if resources were limited or the environmental conditions do not support a high-resolution monitoring programme, low-resolution monitoring could be undertaken to understand the performance of river restoration schemes. The ADCP is very versatile and could be used to expedite the collection of both high and low-resolution monitoring datasets. The findings of this chapter suggest that the comparison of low and high-resolution datasets within a monitoring programme could lead to misleading and spurious results. Therefore, it is recommended that consistency in data resolution within data-driven performance assessments is maintained and that data is resampled if necessary. The temporal resolution of the data collection was also highlighted as an important component influencing the ability to identify geomorphological processes. Therefore, it is recommended that surveys for data-driven performance assessments are captured as frequently as possible (at least bi-annually), for example, to identify seasonal patterns of sediment transport that are potentially ecologically relevant.

Chapter 7 highlighted many advantages of using novel technologies to undertake data-driven performance assessments, such as expedited data collection and health and safety benefits, and presented guidance on which technologies may be suitable for data collection in different situations. However, several barriers were identified in the application of data-driven performance assessments, most notably the time and resources required for data processing and the ability to collapse large datasets into meaningful outputs. Automation and the development of an open-source graphical user interface (GUI) would be critical for supporting the uptake of data-driven performance assessments outside of academia, as it would alleviate some of the technical skills and required time, both of which are costly.

Chapter 7 also addressed the fifth objective of this research which was to "*present guidance for practitioners and the concept of a dissemination framework to facilitate data-driven river restoration monitoring beyond academia".* The data captured in the RHES monitoring programme was found to be excellent for developing non-technical visual dissemination outputs. These were well received by stakeholders. There are a range of possibilities for using this type of data to develop augmented reality tools that were identified within Chapter 7. Chapter 7 suggested that were resources are limited, they are targeted at projects with a strong stakeholder engagement component. This would maximise the promotion of river restoration, a point raised by the International Union for the Conservation of Nature (Addy et al., 2016).

A review of data dissemination methods in river restoration identified that online databases, such as the RiverWiki, and social media may present excellent future opportunities to engage both technical and non-technical audiences (Chapter 2). The RiverWiki presents novel features for interaction amongst the river restoration community as well as a platform sharing reports and datasets. The further advocation and use of this resource by the river restoration community (within the UK and further afield) are recommended to improve knowledge exchange. It was noted that it was difficult to determine whether river restoration schemes had been successful and to identify the continued performance of schemes over time. Consequently, there is a need to develop non-technical performance tracking tools that may help the non-expert understand how schemes are performing over time. This may be particularly useful for funding bodies to track the impact of their investment.

The concept of a framework to facilitate this was presented in Chapter 7. The framework would integrate geomorphological and physical habitat data-driven performance assessments with the monitoring results of other river restoration objectives. This framework would require further development but presents opportunities to guide adaptive management and integrate with existing learning platforms to disseminate project performance to a wide range of stakeholders.

The final outputs of the tool are designed such that they could be disseminated on social media and integrated with online databases such as the RiverWiki. Thus, the tool promotes the performance of river restoration to technical and non-technical stakeholders. It is likely that the tool would need to be complemented by further guidance and training on how to set suitable SMART objectives for river restoration schemes. One limitation of this framework is that it does not facilitate the dissemination of lessons learnt by 'surprise'. The framework is presented for further development and discussion amongst the river restoration community with the aim of improving the learning and adaptive management within river restoration; the overall aim of this research.

8.2 Additional Research findings

This research has primarily focused on data-driven performance assessments but there have been some additional research findings which may be significant for river restoration practice. These are discussed in two sections; first, findings relating to new insights for capturing scientific baseline information for river restoration design and evaluation, and second, findings relating to lessons learned for future spawning habitat restoration.

8.2.1 New insights for capturing scientific baseline information for river restoration design and evaluation

The specific focus of this research was on data-driven performance assessments for river restoration schemes. However, the importance of baseline catchment information, as well as site-specific data, soon became clear. This supports other studies reported in the literature (Wohl, 2005; Downs et al., 2011; Friberg et al., 2016; Beller et al., 2016). This information was critical in the evaluation processes of both geomorphological (Chapter 5) and physical habitat (Chapter 6) performance at the reach scale. Undertaking a review of the River Rother catchment (Chapter 3) demonstrated the wealth of information that has become available through the advent of the internet and willingness to share information in the public domain over the last 20 years. Existing guidance in Sear et al. (2010) presents some useful sources of information, but the range of sources could be updated based on the observations of this research. The sources used within this thesis are largely related to the UK and include;

- Local and national archives;
- Cross-sections from existing flood-risk models;
- Open LiDAR and oblique aerial imagery (emergency response);
- Google Earth™;
- A vision of Britain through time website (http://www.visionofbritain.org.uk);
- Britain from Above website (https://britainfromabove.org.uk);
- Art UK website (https://artuk.org);
- The Mills Archive website (https://millsarchive.org) and
- Geograph website (http://www.geograph.org.uk/).

In addition to baseline information of the catchment, site-specific baseline monitoring was imperative as highlighted by Downs and Kondolf, 2002, Palmer et al., 2005 and England et al., 2008. This research has further highlighted that baseline monitoring is important for identifying suitable geomorphological and physical habitat objectives and specific design criteria. Had an adequate period of baseline monitoring been undertaken for this project in sufficient time to inform design, the resulting design may have been different. For example, the potential for deposition of sediment over the riffle feature that was identified within the dataset may have been identified prior to construction and resulted in a modified design (Chapter 5).

A significant limitation of the RHES monitoring programme conducted here as part of this research was the lack of baseline temporal monitoring data over a range of flows. A single low flow survey was not adequate to fully evaluate the geomorphological and physical habitat performance of the RHES. Additionally, the short lead in time to the monitoring programme limited the opportunity to capture a ‘control’ site. Had this been possible, some uncertainty surrounding the geomorphological effects of the 2013/14 flood events may have been reduced. Nonetheless, valuable lessons for spawning habitat restoration practices have been learnt which are discussed in the following section.

By recognising the shortcomings of this research study, recommendations for best-practice baseline monitoring were made. A minimum period of 2 years of baseline monitoring is recommended to maximise the opportunity for gathering sufficient data (Chapter 7). However, this led to an interesting discussion on the reality of baseline monitoring, as funding is not always available for this activity. Consequently, without further education of funding bodies on the value of baseline monitoring, it may perhaps still be largely an academic pursuit. There is a risk that river restoration monitoring may become a tick-box exercise within current funding applications. Chapter 7 suggested that funding bodies could play a greater role in promoting and reviewing river restoration monitoring. There could be scope for a seed fund to support the baseline monitoring and design phases of river restoration. Full river restoration projects could be funded subject to certain requirements being met (e.g. appropriate baseline monitoring and designs).

8.2.2 Lessons learned for future spawning habitat restoration

The exploratory programme of the RHES indicated that, given the challenging conditions presented within the initial 18-month period, the scheme was largely successful in achieving improved hydraulic physical habitat for spawning brown trout. However, sediment infiltration is a concern for the riffle feature and further monitoring is recommended but there is again uncertainty on how this will be funded (Chapters 6 and 7). From the initial period of monitoring, despite an imperfect baseline data set, some important research findings were gleaned which may inform spawning habitat restoration practice.

First, the monitoring programme supports research that highlights the importance of considering catchment-scale processes when undertaking spawning habitat restoration (Wheaton et al., 2004). Deposition of material over the riffle feature in combination with observations by Evans et al. (2017) indicates that the gravel was not fully self-cleansing (Chapter 5). Importantly, the fine sediment accumulation noted over the riffle feature was not apparent from the surface, and therefore it is a process that may be underestimated by citizen science visual identification surveys. Second, the longevity of the RHES riffle feature may be limited if further gravel augmentation is not undertaken within the reach. Based on this observation, where fine sediment accumulation is a significant risk, other measures (e.g. deflectors) are recommended to promote sediment transport and increase the longevity of the augmented features. Furthermore, if the longitudinal connectivity of sediment transport (particularly for gravel) is disrupted, regular gravel augmentation is recommended to maintain constructed riffle features and restore natural sediment transport processes as far as possible.

Third, the design of the RHES riffle feature was somewhat experimental with the crest constructed over the tail of the riffle feature. During the significant flow events of the 2013/14 floods, the sediments of the riffle feature were reworked such that the riffle crest became sited over the head of the riffle feature (Chapter 5). This is suspected to be a result of a localised channel widening over the head of the riffle feature. However, this is an interesting finding for spawning habitat rehabilitation as it suggests that given the appropriate substrate and range of flows, rivers may create self-sustaining forms. Therefore, resources could potentially be saved during the construction phase of some spawning habitat restoration schemes. This may be a more long-term restoration strategy as significant formative flows may take an extended period to recur. However, there may be a potential to augment flows in some situations, such as if the scheme is sited downstream of a dam or reservoir and the flows could be released without causing undesirable flooding downstream.

Fourth, the monitoring results of the RHES riffle feature also support previous research based on modelled restoration designs by Pasternack and Brown (2013), that suggest a smoother, more gradual morphology of a riffle crest may generally present a more desirable physical habitat. Fifth, this research highlights the importance of considering the spatial configuration of physical habitats for different life stages (Chapter 6). The RHES did consider this but evidence from the literature indicates there may have been inadequate marginal refuge habitat within an appropriate migratory distance to support brown trout through the life cycle as they emerged from the spawning habitat over the riffle feature. Without adequately considering the full life cycle within physical habitat restoration design, spawning habitat restoration may not fully contribute to an improvement in ecological status.

8.3 Future research and development

As with any research, a number of key areas have been identified for future study and development. Limitations on funding and resources for monitoring are issues consistently raised by river restoration practitioners. This research has reiterated the need for strategic allocation of resources for river restoration monitoring. However, for this to be feasible, a review of existing monitoring datasets globally is imperative for identifying which types of schemes are priorities for data-driven performance assessments.

Further high-resolution monitoring of the RHES is recommended to assess the longevity of the scheme considering the very challenging catchment conditions. However, there are limitations of this monitoring study with respect to understanding the performance of the riffle feature in moderate and higher flows compared baseline condition. Consequently, restoration projects using similar schemes should still be worthy of a data-driven performance assessment. The baseline catchment information was critical in interpreting the results of the RHES monitoring programme and this research has identified potential open resources that could be beneficial to other studies. However, these resources are largely UK based and national reviews of similar sources for other countries may be beneficial.

This research identified that the use of fragmentation metrics in conjunction with physical habitat simulations were highly influential on the interpretation of performance. It is generally assumed that habitat quality declines as fragmentation increases. Nonetheless, there appear to be limited studies relating to freshwater habitats that have investigated the ecological relevance of this assumption. Therefore, this may be a future avenue for research. The analysis of the high-resolution data from the ADCP using these metrics as well as general data processing was highly resource intensive and may present a significant resource constraint. Without further development of improved tools to process and analyse the data from intensive monitoring programmes, this type of monitoring is likely to remain largely an academic pursuit.

Where data-driven performance assessments are undertaken, the wide-spread dissemination of project results and datasets are critical to the much-needed continual learning in river restoration. A concept of a performance tracking tool which could potentially integrate with existing learning platforms is presented within this thesis. However, this is still largely a concept and further development and testing with end users is critical. Unless the performance of river restoration schemes is tracked, the potential for learning is likely to remain impeded and river restoration practice will not develop. At worst, the lack of demonstrable performance may risk future funding for river restoration and our rivers may continue to degrade, particularly if they are restored using ineffective techniques.

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Appendix:

A. Physical Habitat Suitability Curves for brown trout, dace and roach.

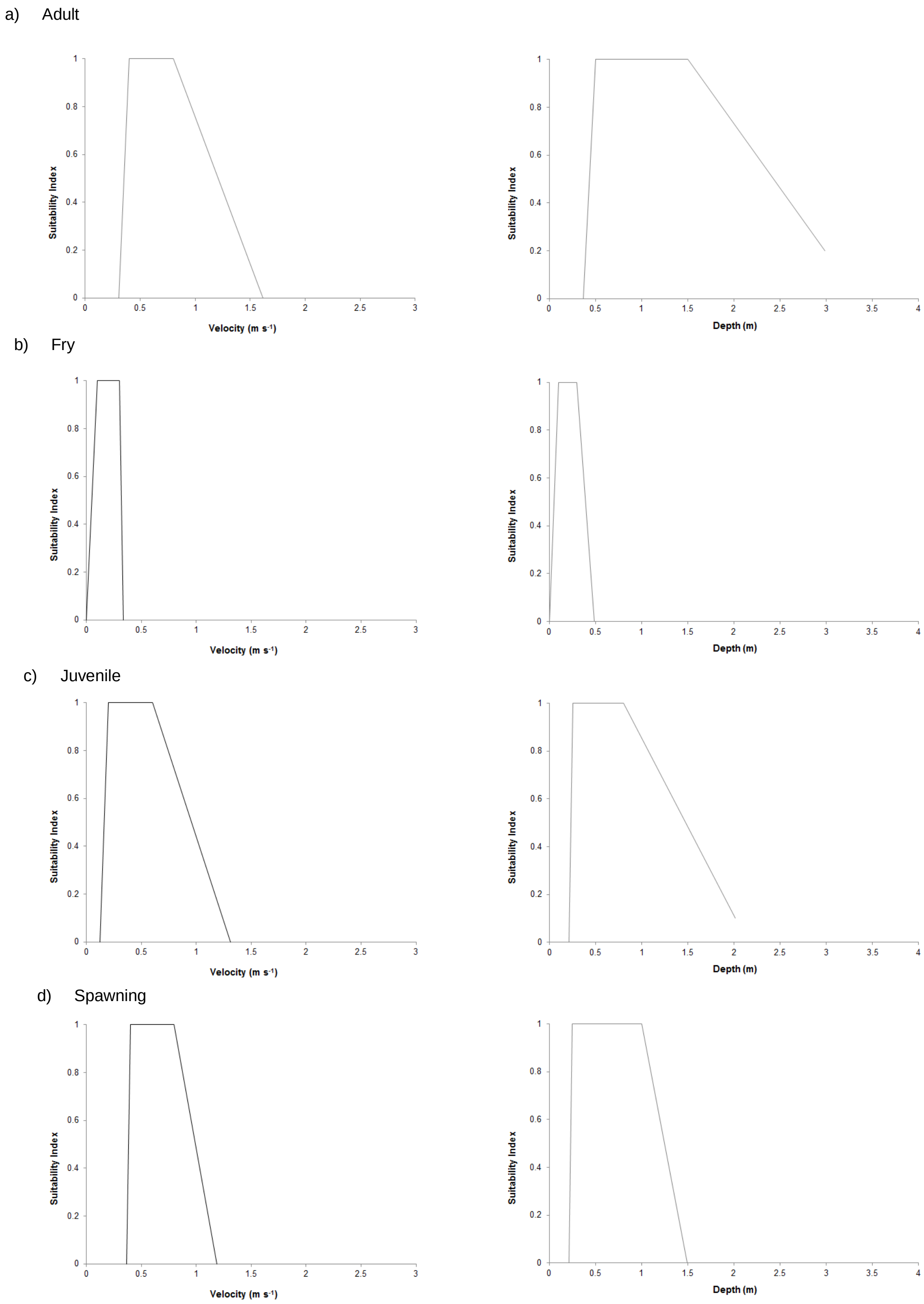
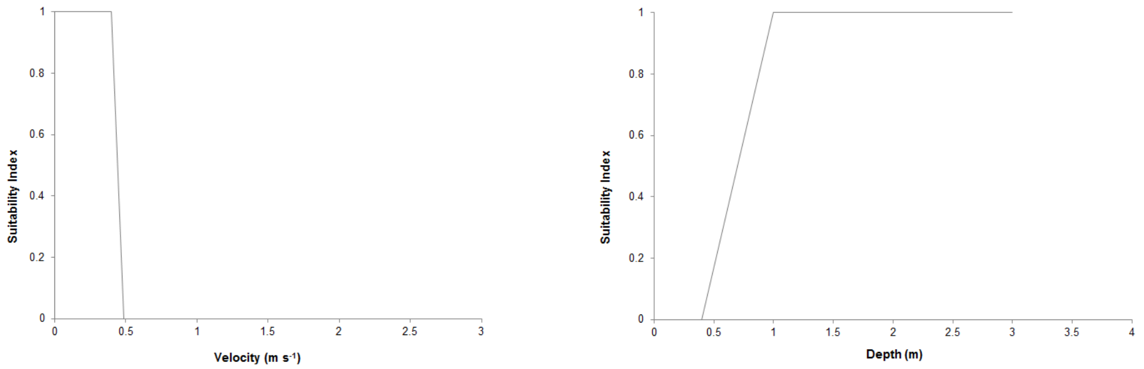
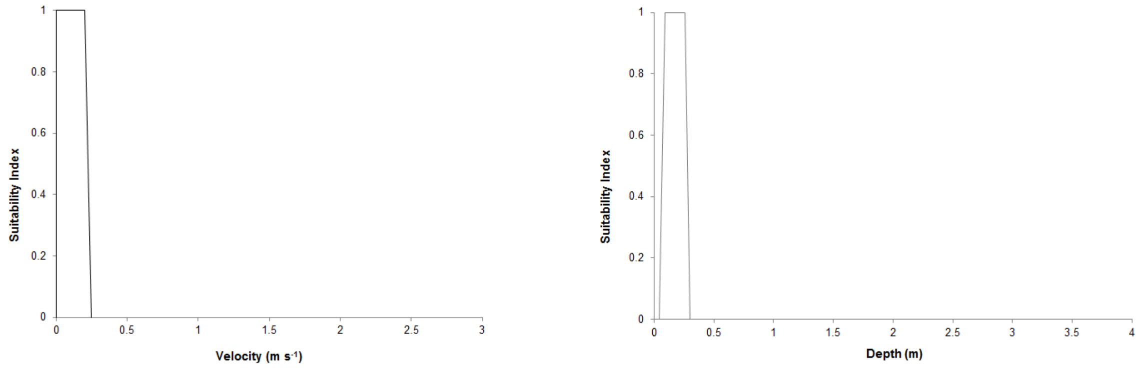


Figure A.1 Habitat Suitability criteria for Brown Trout

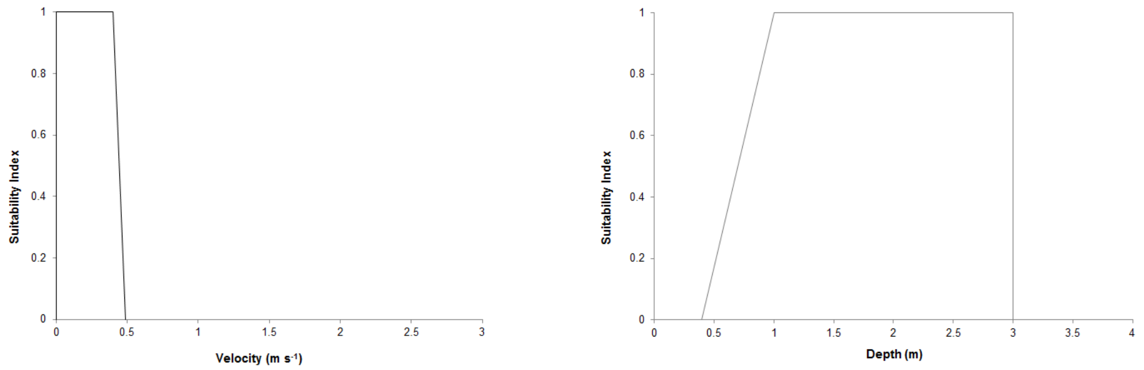
a) Adult



b) Fry



c) Juvenile



d) Spawning

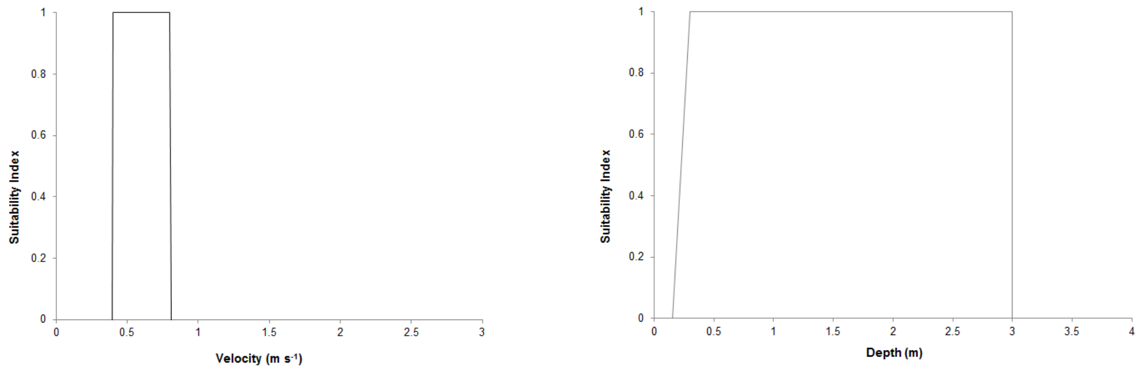
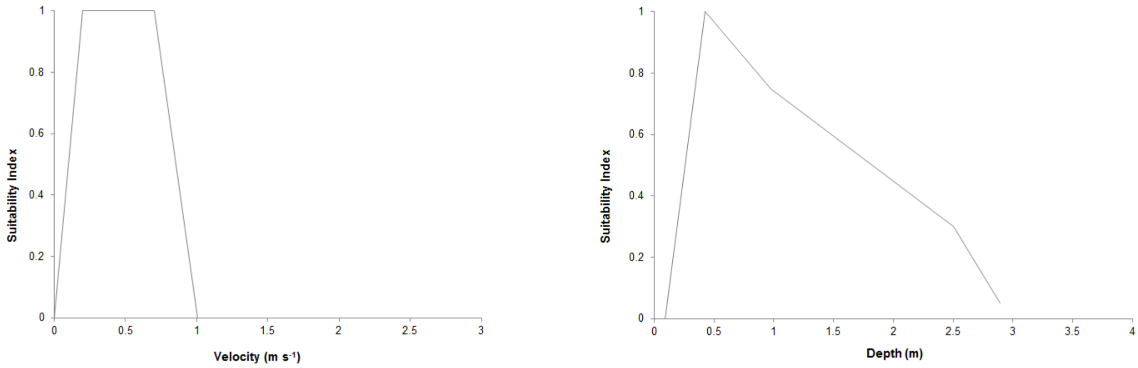
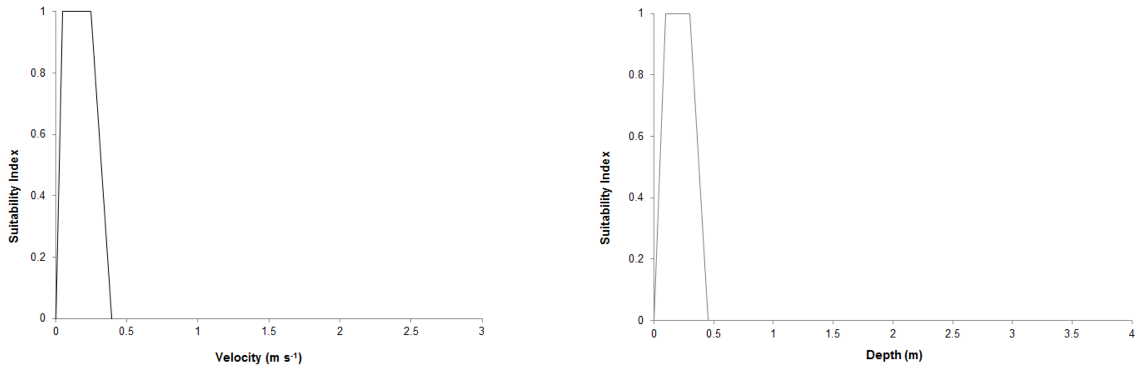


Figure A.2 Habitat Suitability criteria for Roach

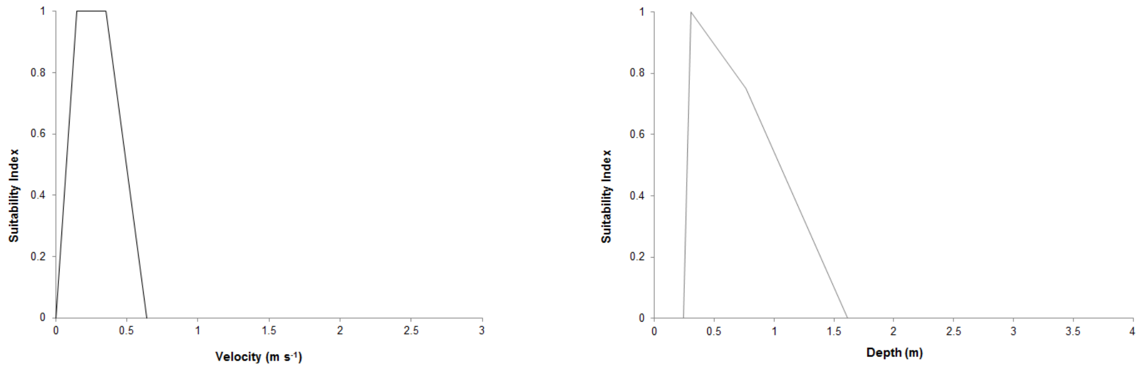
a) Adult



b) Fry



c) Juvenile



d) Spawning

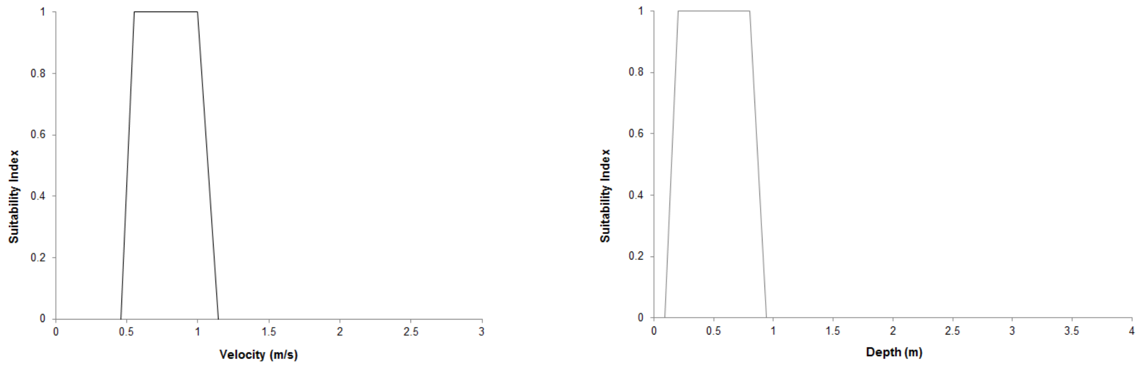


Figure A.3 Habitat Suitability criteria for Dace

Appendix:

B. Summary statistics for elevation and velocity data

Table B.1 Table of weighted summary statistics of elevations over the reach and riffle feature.

Elevations over the Reach (m AOD)

	Mean	Min	Max	Std	Cov	QCD	Median	2.5th P	25th P	75th P	95th P	IQR	Range	Var	Kurt	Skew	Rug
Baseline	3.83	2.43	5.48	0.45	0.12	0.06	3.85	2.94	3.64	4.08	4.57	0.45	3.05	0.21	3.96	-0.23	1.047
As-built	4.00	2.43	5.48	0.53	0.13	0.08	4.08	2.94	3.72	4.41	4.64	0.69	3.05	0.28	3.03	-0.65	1.034
1 Month	3.99	2.42	5.48	0.53	0.13	0.09	4.07	2.93	3.69	4.40	4.67	0.71	3.05	0.28	3.01	-0.60	1.034
3 Months	4.05	2.35	5.48	0.58	0.14	0.11	4.11	2.93	3.62	4.49	4.86	0.87	3.13	0.34	2.78	-0.50	1.035
7 Months	3.85	1.92	5.32	0.72	0.19	0.14	3.99	2.50	3.35	4.44	4.76	1.09	3.40	0.52	2.46	-0.59	1.035
12 Months	4.11	2.30	5.31	0.70	0.17	0.14	4.31	2.77	3.58	4.71	4.94	1.13	3.01	0.50	2.31	-0.67	1.032
18 Months	3.92	1.85	5.54	0.77	0.20	0.14	4.08	2.34	3.45	4.54	4.82	1.09	3.69	0.59	2.79	-0.77	1.034

Elevations over the Riffle area (m AOD)

	Mean	Min	Max	Std	Cov	QCD	Median	2.5th P	25th P	75th P	95th P	IQR	Range	Var	Kurt	Skew	Rug
Baseline	3.97	3.44	5.48	0.34	0.09	0.04	3.90	3.58	3.76	4.09	4.65	0.33	2.04	0.12	6.17	1.59	1.0612
As-built	4.44	3.88	5.48	0.21	0.05	0.02	4.44	4.11	4.35	4.54	4.74	0.19	1.60	0.04	7.00	1.01	1.0220
1 Month	4.43	3.85	5.48	0.21	0.05	0.02	4.43	4.10	4.32	4.52	4.75	0.20	1.63	0.05	6.68	1.02	1.0203
3 Months	4.53	4.11	5.48	0.21	0.05	0.02	4.50	4.25	4.41	4.63	4.94	0.22	1.36	0.04	4.94	1.08	1.0212
7 Months	4.52	3.99	5.26	0.20	0.04	0.03	4.49	4.24	4.38	4.63	4.87	0.25	1.27	0.04	4.09	0.70	1.0203
12 Months	4.76	4.28	5.31	0.15	0.03	0.02	4.75	4.52	4.67	4.87	4.99	0.20	1.03	0.02	3.17	-0.16	1.0196
18 Months	4.61	4.08	5.34	0.18	0.04	0.02	4.59	4.36	4.50	4.72	4.91	0.23	1.25	0.03	4.59	0.60	1.0221

Table B.2 Table of weighted summary statistics of velocity (40% slice) over the reach and riffle feature.

Survey	Minimum (V_{min})	Minimum (V_{min})	Weighted Mean (V_{μ})	Weighted Coefficient of Variation (V_{CV})	Weighted Skewness (V_s)	Weighted Kurtosis (V_k)	Weighted IQR (V_{IQR})	Weighted Median (V_{50})
Baseline								
Reach	0.00	0.54	0.19	0.66	0.35	2.09	0.22	0.17
Riffle	0.00	0.43	0.22	0.60	-0.12	1.63	0.24	0.23
As-built								
Reach	0.00	1.15	0.26	0.82	1.39	5.22	0.26	0.26
Riffle	0.00	1.15	0.44	0.57	0.54	3.17	0.27	0.43
1 Month								
Reach	0.00	1.24	0.25	0.86	1.48	5.70	0.28	0.19
Riffle	0.00	1.24	0.43	0.57	0.74	3.68	0.27	0.42
3 Months								
Reach	0.00	1.01	0.44	0.68	0.04	1.65	0.58	0.47
Riffle	0.00	1.01	0.61	0.49	-0.67	2.08	0.50	0.71
7 Months								
Reach	0.00	0.98	0.47	0.61	-0.05	1.70	0.53	0.48
Riffle	0.00	0.98	0.63	0.43	-0.86	2.45	0.40	0.74
12 Months								
Reach	0.00	0.84	0.25	0.90	0.80	2.40	0.34	0.17
Riffle	0.00	0.84	0.45	0.50	-0.59	2.22	0.33	0.52
18 Months								
Reach	0.00	1.09	0.44	0.70	0.36	1.73	0.58	0.38
Riffle	0.00	1.09	0.67	0.44	-0.83	2.34	0.43	0.80

Appendix:

C. Physical Habitat Suitability testing using high and low-resolution data

Table C.1 Physical habitat simulation results for brown trout using high and low-resolution data

	Total Suitable Area	Low Quality				Moderate Quality				High Quality			
		% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension
Adult brown trout													
As-built survey													
High resolution	389	9	5	1.8	1.6	25	10	1.9	1.7	66	5	0.8	1.4
Low resolution	248	4	4	3.4	1.8	13	4	2.1	1.6	83	3	0.6	1.3
12-month survey													
High resolution	423	8	5	2.4	1.7	10	6	2.0	1.6	83	3	0.7	1.4
Low resolution	465	3	3	1.9	1.5	15	5	1.2	1.4	82	3	0.4	1.3
Juvenile brown trout													
As-built survey													
High resolution	1008	1	6	4.9	2.2	11	9	1.5	1.6	88	3	0.4	1.3
Low resolution	939	2	4	2.2	1.6	10	7	1.4	1.5	87	1	0.3	1.3
12-month survey													
High resolution	926	3	7	3.0	1.8	15	11	1.3	1.5	82	4	0.3	1.3
Low resolution	1025	4	4	1.7	1.5	14	8	1.2	1.5	83	3	0.3	1.2

Table C.1 (Continued) Physical habitat simulation results for brown trout using high and low-resolution data

	Total Suitable Area	Low Quality				Moderate Quality				High Quality			
		% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension
Fry brown trout													
As-built survey													
High resolution	70	29	10	3.5	1.9	25	10	4.1	2.0	46	11	3.5	1.9
Low resolution	80	4	3	4.7	2.3	33	9	3.1	1.9	63	14	3.5	1.9
12-month survey													
High resolution	192	43	20	2.8	1.8	35	16	2.9	1.8	22	12	3.8	2.0
Low resolution	155	35	13	2.5	1.8	29	12	3.3	1.9	36	9	2.9	1.8
Spawning brown trout													
As-built survey													
High resolution	360	1	2	6.1	3.7	8	2	1.5	1.4	92	3	0.6	1.3
Low resolution	256	1	1	3.1	1.4	1	1	2.2	1.0	98	1	0.4	1.2
12-month survey													
High resolution	511	1	2	3.5	1.8	2	3	2.8	1.7	97	2	0.4	1.2
Low resolution	435	1	1	2.6	1.1	1	3	3.3	1.8	98	2	0.3	1.2

Table C.2 Physical habitat simulation results for dace using high and low-resolution data

		Low Quality				Moderate Quality				High Quality			
	Total Suitable Area	% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension
Adult dace													
As-built survey													
High resolution	1374	19	15	1.14	1.55	24	12	1.50	1.66	57	2	0.48	1.36
Low resolution	1444	16	11	0.83	1.42	38	5	0.80	1.49	46	1	0.44	1.32
12-month survey													
High resolution	1763	33	9	0.79	1.49	23	12	1.07	1.56	44	3	0.43	1.33
Low resolution	1818	31	10	0.64	1.42	24	9	1.04	1.56	45	2	0.34	1.26
Juvenile dace													
As-built survey													
High resolution	1283	22	12	1.20	1.57	39	11	0.95	1.54	40	6	0.64	1.41
Low resolution	1238	20	8	0.88	1.45	43	5	0.70	1.44	38	3	0.65	1.41
12-month survey													
High resolution	1263	41	13	1.13	1.60	44	13	0.86	1.51	14	9	1.85	1.70
Low resolution	1284	37	11	0.91	1.52	43	8	0.93	1.54	20	6	1.35	1.61

Table C.2 (Continued) Physical habitat simulation results for dace using high and low-resolution data

	Total Suitable Area	Low Quality				Moderate Quality				High Quality			
		% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension
Fry dace													
As-built survey													
High resolution	60	22	7	3.78	1.96	30	9	3.85	1.97	48	12	3.91	1.99
Low resolution	73	4	3	5.50	2.67	21	8	4.23	2.04	76	15	3.90	1.99
12-month survey													
High resolution	150	37	19	3.26	1.90	31	17	3.87	1.98	32	14	4.01	2.00
Low resolution	143	26	10	3.15	1.87	35	14	3.07	1.86	39	9	3.25	1.90
Spawning dace													
As-built survey													
High resolution	262	3	3	3.60	1.90	15	5	1.76	1.55	82	2	0.50	1.23
Low resolution	168	2	2	3.43	1.74	4	2	2.39	1.46	94	1	0.46	1.14
12-month survey													
High resolution	383	-	-	-	-	-	-	-	-	97	1	0.41	1.23
Low resolution	262	1	1	3.16	1.34	1	1	3.13	1.33	98	1	0.44	1.20

Table C.3 Physical habitat simulation results for roach using high and low-resolution data

		Low Quality				Moderate Quality				High Quality			
	Total Suitable Area	% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension
Adult roach													
As-built survey													
High resolution	1188	8	10	1.91	1.67	17	8	1.02	1.48	75	4	0.36	1.29
Low resolution	1076	5	10	2.13	1.68	11	13	1.83	1.67	84	2	0.32	1.26
12-month survey													
High resolution	1240	7	14	2.13	1.71	12.41	13	1.72	1.67	81	3	0.29	1.24
Low resolution	1352	8	13	2.20	1.74	6	21	3.30	1.91	86	2	0.25	1.22
Juvenile roach													
As-built survey													
High resolution	1188	8	10	1.91	1.67	17	8	1.02	1.48	75	4	0.36	1.29
Low resolution	1076	5	10	2.13	1.68	11	13	1.83	1.67	84	2	0.32	1.26
12-month survey													
High resolution	1240	7	14	2.13	1.71	12	13	1.72	1.67	81	3	0.29	1.24
Low resolution	1352	8	13	2.20	1.74	6	21	3.30	1.91	86	2	0.25	1.22

Table C.3 (Continued) Physical habitat simulation results for roach using high and low-resolution data

		Low Quality				Moderate Quality				High Quality			
	Total Suitable Area	% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension
Fry roach													
As-built survey													
High resolution	9	-	-	-	-	-	-	-	-	100	7	5.15	2.23
Low resolution	11	-	-	-	-	2	1	12.17	0.44	98	11	6.69	2.43
12-month survey													
High resolution	20	-	-	-	-	-	-	-	-	100	15	7.59	2.43
Low resolution	30	-	-	-	-	-	-	-	-	100	12	5.19	2.15
Spawning roach													
As-built survey													
High resolution	0	-	-	-	-	-	-	-	-	-	-	-	-
Low resolution	202	-	-	-	-	-	-	-	-	100	1	0.59	1.28
12-month survey													
High resolution	20	-	-	-	-	-	-	-	-	100	15	7.59	2.43
Low resolution	0	-	-	-	-	-	-	-	-	-	-	-	-

Appendix:

D. Physical Habitat Suitability testing using different representations of velocity

Table D.1 Physical habitat simulation results for brown trout using high and low-resolution data

	Total Suitable Area	Low Quality				Moderate Quality				High Quality			
		% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension
Adult brown trout													
20% depth velocity	265	13	10	2.96	1.83	31	8	1.83	1.65	56	5	1.06	1.47
40% depth velocity	375	14	8	2.00	1.65	28	10	1.98	1.70	58	5	0.86	1.43
5 point av. depth velocity	423	19	11	2.18	1.72	30	10	1.81	1.67	51	5	0.86	1.43
20% & 80% av. depth velocity	409	14	12	2.62	1.79	33.54	9	1.67	1.65	53	5	0.88	1.44
Juvenile brown trout													
20% depth velocity	927	0	2	4.64	2.55	10	13	1.81	1.65	89	3	0.40	1.31
40% depth velocity	993	1	6	4.40	2.09	10	9	1.41	1.54	89	3	0.41	1.33
5 point av. depth velocity	1077	5	7	1.97	1.65	18	7	1.16	1.53	77	1	0.40	1.31
20% & 80% av. depth velocity	1037	2	7	3.10	1.83	13	17	1.81	1.68	85	4	0.38	1.31

Table D1 (Continued) Physical habitat simulation results for brown trout using high and low-resolution data

	Total Suitable Area	Low Quality				Moderate Quality				High Quality			
		% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension
Fry brown trout													
20% depth velocity	117	26	13	3.86	1.98	30	9	2.59	1.76	44	12	3.54	1.94
40% depth velocity	100	25	12	4.15	2.02	34	7	2.75	1.79	41	13	3.31	1.90
5 point av. depth velocity	101	32	11	3.60	1.94	38	12	2.81	1.81	30	7	3.28	1.88
20% & 80% av. depth velocity	106	24	12	4.00	2.00	33	12	3.24	1.88	43	13	3.73	1.96
Spawning brown trout													
20% depth velocity	275	-	-	-	-	1	3	6.49	3.03	99	2	0.53	1.28
40% depth velocity	359	-	-	-	-	9	2	1.28	1.34	91	3	0.59	1.34
5 point av. depth velocity	399	3	5	4.05	2.01	13	6	1.92	1.63	84	2	0.59	1.34
20% & 80% av. depth velocity	399	2	5	4.05	2.01	6	7	2.94	1.81	97	2	0.49	1.29

Table D.2 Physical habitat simulation results for dace using high and low-resolution data

		Low Quality				Moderate Quality				High Quality			
	Total Suitable Area	% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension
Adult dace													
20% depth velocity	1372	19	20	1.21	1.57	30	8	1.21	1.60	51	3	0.50	1.36
40% depth velocity	1372	18	16	1.15	1.55	25	11	1.46	1.65	57	2	0.48	1.36
5 point av. depth velocity	1384	21	15	1.10	1.54	26	10	1.28	1.61	53	4	0.44	1.33
20% & 80% av. depth velocity	1404	17	16	1.16	1.55	29	10	1.10	1.57	54	3	0.44	1.33
Juvenile dace													
20% depth velocity	1285	21	11	1.12	1.55	32	10	1.12	1.58	46	3	0.53	1.37
40% depth velocity	1252	20	12	1.26	1.58	38	11	1.03	1.56	41	5	0.63	1.41
5 point av. depth velocity	1305	25	11	1.11	1.56	41	11	0.85	1.51	33	7	0.65	1.40
20% & 80% av. depth velocity	1281	22	12	1.28	1.60	35	11	1.02	1.55	42	4	0.63	1.42

Table D2 (Continued) Physical habitat simulation results for dace using high and low-resolution data

		Low Quality				**Moderate Quality**				**High Quality**			
	Total Suitable Area	% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension
Fry dace													
20% depth velocity	90	23	11	4.07	2.01	24	9	3.62	1.93	54	8	3.48	1.93
40% depth velocity	76	19	8	4.43	2.08	33	9	3.30	1.88	48	10	3.49	1.92
5 point av. depth velocity	87	35	12	3.51	1.92	29	10	3.46	1.91	36	12	3.69	1.95
20% & 80% av. depth velocity	86	22	10	4.05	2.01	32	11	3.41	1.90	46	12	3.72	1.96
Spawning dace													
20% depth velocity	129	10	4	3.41	1.87	7	2	2.55	1.58	84	2	0.68	1.24
40% depth velocity	263	4	3	3.31	1.83	15	5	1.76	1.55	82	2	0.50	1.23
5 point av. depth velocity	297	9	3	1.79	1.51	13	3	1.75	1.55	78	1	0.52	1.25
20% & 80% av. depth velocity	293	-	-	-	-	-	-	-	-	80	1	0.45	1.20

Table D.3 Physical habitat simulation results for roach using high and low-resolution data

	Total Suitable Area	Low Quality				Moderate Quality				High Quality			
		% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension
Adult roach													
20% depth velocity	1239	12	7	1.29	1.55	18	8	1.01	1.49	70	5	0.35	1.28
40% depth velocity	1163	10	9	1.58	1.60	17	7	1.02	1.48	73	7	0.34	1.27
5 point av. depth velocity	1286	18	6	0.92	1.46	16	7	0.98	1.47	66	5	0.33	1.26
20% & 80% av. depth velocity	1253	14	6	1.10	1.50	17.40	7	0.87	1.44	68	4	0.33	1.26
Juvenile roach													
20% depth velocity	1239	12	7	1.29	1.55	18	8	1.01	1.49	70	5	0.35	1.28
40% depth velocity	1163	10	9	1.58	1.60	17	7	1.02	1.48	73	7	0.34	1.27
5 point av. depth velocity	1286	18	6	0.92	1.46	16	7	0.98	1.47	66	5	0.33	1.26
20% & 80% av. depth velocity	1253	14	6	1.10	1.50	17	7	0.87	1.44	68	4	0.33	1.26

Table D3 (Continued) Physical habitat simulation results for roach using high and low-resolution data

	Total Suitable Area	Low Quality				Moderate Quality				High Quality			
		% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension	% Area	Divisions	Edge Ratio	Fractal Dimension
Fry roach													
20% depth velocity	17	-	-	-	-	-	-	-	-	100	8	4.76	2.12
40% depth velocity	14	-	-	-	-	-	-	-	-	100	6	4.43	2.08
5 point av. depth velocity	68	80	2	1.27	1.42	2	1	6.26	3.71	18	6	4.71	2.13
20% & 80% av. depth velocity	14	-	-	-	-	-	-	-	-	100	7	4.71	2.12
Spawning roach													
20% depth velocity	244	-	-	-	-	-	-	-	-	100	2	0.63	1.33
40% depth velocity	268	-	-	-	-	-	-	-	-	100	4	0.69	1.37
5 point av. depth velocity	363	4	4	3.27	1.82	19	3	0.97	1.33	77	3	0.66	1.36
20% & 80% av. depth velocity	984	1	6	1.09	1.41	16	15	1.75	1.67	83	6	0.72	1.49

Appendix:

E. Demonstration of the performance tracking framework concept – example of integration of different river restoration objectives.

Please note that this scenario is *fictional, intended only* for demonstration purposes and thus ***should not*** be used to judge the performance of the RHES. Four objectives are defined to demonstrate the versatility of the concept; two physical habitats, one ecological and one social. The physical habitat and ecological objectives are evaluated using real data, however, no social data was available (nor an objective of the scheme) and is hypothetical. For each objective; the indicator of performance is highlighted in blue, the ideal target is highlighted in green, the acceptable target is highlighted in pink.

Objective 1 (outlined in Box E.1) demonstrates how the framework may guide objectives that may become less important in the medium (5-10 years) and longer (more than 10 years) timeframes. It also establishes a target range between 'success' and 'further intervention' that highlights that further monitoring is recommended. This staged approach accounts for a degree of uncertainty within the results. Objective 2 (outlined in Box E.2) demonstrates that the importance of an objective may not change over time and that objectives do not necessarily need to be staged. There may be a requirement of a scheme to meet certain criteria by a deadline, and further intervention may need to be undertaken to achieve this.

Objective 3 (outlined in Box E.4) demonstrates that an objective may become more important over time. In this example, the objective is ecological, if in the first 5 years if no improvement is seen no further recommendation will be suggested. In the longer-term objectives, the further recommendation will be suggested if a target isn't met. In addition, the importance of this objective (through weighting) increases over time. In this hypothetical example, this may account for a delay in the recovery of a species population. Objective 4 (outlined in Box E.5) further demonstrates that this framework may be used to track the performance of a range different types of restoration objectives. Additionally, it highlights that multiple indicators may be used to evaluate the success of an objective and that creating a band of uncertainty may not be necessary for each objective.

Box E.1: Objective 1 - Physical Habitat

To improve the abundance of suitable physical habitat for spawning brown trout. This will be measured by monitoring the total suitable area given as defined by preference curves for this species. Success will be defined as:

Short term (less than 5 years post-restoration):

Weighting = 40 %

- At least a 100% increase (230 m^2) of suitable habitat provisioned at low flows on the baseline condition, an 80% increase (207 m^2) would be sufficient increase to not recommend further intervention.
- To provide a minimum of 400 m^2 of suitable habitat at moderate flows, a 15 % decrease (340 m^2) will be a sufficient increase to not recommend further intervention.
- To provide a minimum of 300 m^2 of suitable habitat at high flows, a 10 % decrease (270 m^2) will be a sufficient increase to not recommend further intervention.

Medium term (5-10 years post-restoration):

Weighting = 30 %

- At least a 100% increase (230 m^2) of suitable habitat provisioned at low flows on the baseline condition, a 70 % increase (196 m^2) would be sufficient increase to not recommend further intervention.
- To provide a minimum of 400 m^2 of suitable habitat at moderate flows, a 20 % (320 m^2) decrease will be a sufficient increase to not allow intervention.
- To provide a minimum of 300 m^2 of suitable habitat at high flows, a 20 % (240 m^2) decrease will be a sufficient increase to not allow intervention.

Long term (more than 10 years post-restoration):

Weighting = 25 %

- At least a 100 % increase (230 m^2) of suitable habitat provisioned at low flows on the baseline condition, a 60 % increase (184 m^2) would be sufficient increase to not recommend further intervention.
- To provide a minimum of 400m^2 of suitable habitat at moderate flows, a 30% (280 m^2) decrease will be a sufficient increase to not allow intervention.

- To provide a minimum of 300m^2 of suitable habitat at high flows, a 30% (210 m^2) decrease will be a sufficient increase to not allow intervention.

Box E.2: Objective 2 - Physical Habitat

To maintain the abundance of suitable physical habitat for adult dace. This will be measured by monitoring the total suitable area given as defined by preference curves for this species. Success will be defined as:

Short term (less than 5 years post-restoration):

Weighting = 10 %

- To maintain the abundance of suitable habitat provisioned at low flows found during the baseline condition (1338 m^2), a 20% decrease (1070 m^2) would be sufficient to not recommend further intervention.
- To maintain a minimum abundance of 2000 m^2 of suitable habitat at moderate flows, a 20% decrease (1600 m^2) would be sufficient to not recommend further intervention.
- To maintain a minimum abundance of 2000 m^2 of suitable habitat at high flows, a 20% decrease (1600 m^2) would be sufficient to not recommend further intervention.

Medium term (5-10 years post-restoration):

Weighting = 10 %

- To maintain the abundance of suitable habitat provisioned at low flows found during the baseline condition (1338 m^2), a 10% decrease (1204 m^2) would be sufficient to not recommend further intervention.
- To maintain a minimum abundance of 2000 m^2 of suitable habitat at moderate flows, a 10% decrease (1600 m^2) would be sufficient to not recommend further intervention.
- To maintain a minimum abundance of 2000 m^2 of suitable habitat at high flows, a 10% decrease (1600 m^2) would be sufficient to not recommend further intervention.

Long term (more than 10 years post-restoration):

Weighting = 10 %

- To maintain the abundance of suitable habitat provisioned at low flows found during the baseline condition (1338 m^2), any decrease in this value will result in a recommendation for further intervention.

- To maintain a minimum abundance of 2000 m^2 of suitable habitat at moderate flows, any decrease in this value will result in a recommendation for further intervention.
- To maintain a minimum abundance of 2000 m^2 of suitable habitat at high flows, any decrease in this value will result in a recommendation for further intervention.

Box E.3: Objective 3 - Ecological

To improve the number of brown trout within the lower catchment of the River Rother. This will be evaluated using the density values from annual fish monitoring report, using the Coultershaw Bridge as a representative site. Success will be defined as:

Short term (less than 5 years post-restoration):

Weighting = 10 %

- To maintain the density of fish (individuals per $100m^2$) seen prior to restoration as seen prior to restoration, further monitoring is recommended if the density is less than 1.

Medium term (5-10 years post-restoration):

Weighting = 30 %

- To improve the density of fish by to at least 2 individuals per $100m^2$ on the baseline conditions, an improvement in density to 1 individual per $100m^2$ would be sufficient to not recommend further intervention.

Long term (more than 10 years post-restoration):

Weighting = 50 %

- To improve the density of fish by at least 3 (individuals per $100m^2$) on the baseline conditions, an improvement in density to 2 individuals per $100m^2$ would be sufficient to not recommend further intervention.

Box E.4: Objective 4 - Social

To demonstrate the benefit of the river restoration scheme to local community groups, these may include schools, universities, angling clubs, rambling clubs and local businesses. The success will be measured by feedback forms from the local community groups after the event. Success will be defined as:

Short term (less than 5 years post-restoration):

Weighting = 40 %

- To deliver at least 3 community engagement events per year to a range of stakeholders and receive at least 60 % positive feedback from these events.

Medium term (5-10 years post-restoration):

Weighting = 20 %

- To deliver at least 2 community engagement events per year to a range of stakeholders and receive at least 75 % positive feedback from these events.

Long term (more than 10 years post-restoration):

Weighting = 15 %

- To deliver at least 2 community engagement events per year to a range of stakeholders and receive at least 95 % positive feedback from these events.

The short-term results of this hypothetical scenario are plotted on a series of graphs and summarised within a pie chart (Figure E.1). The results indicate the following over the short-term;

- The scheme is not performing relative to the first physical habitat objective of improving spawning habitat for brown trout at high flows. Consequently, *further intervention* is recommended for this objective.
- The scheme is performing relative to the second physical habitat objective of maintaining habitat for adult dace at all high flows. Consequently, this performance relative to this objective is suggested as *successful*.

- There is uncertainty around the third objective of improving the density of brown trout and the target has not been met. Consequently, *further monitoring* is recommended to establish performance relative to this objective.
- The scheme is performing relative to the fourth objective of demonstrating the benefit of river restoration to stakeholders. Consequently, this performance relative to this objective is suggested as *successful*.

Overall, the scheme had some successes over the short-term but further intervention may be required to meet the main objective of the scheme.

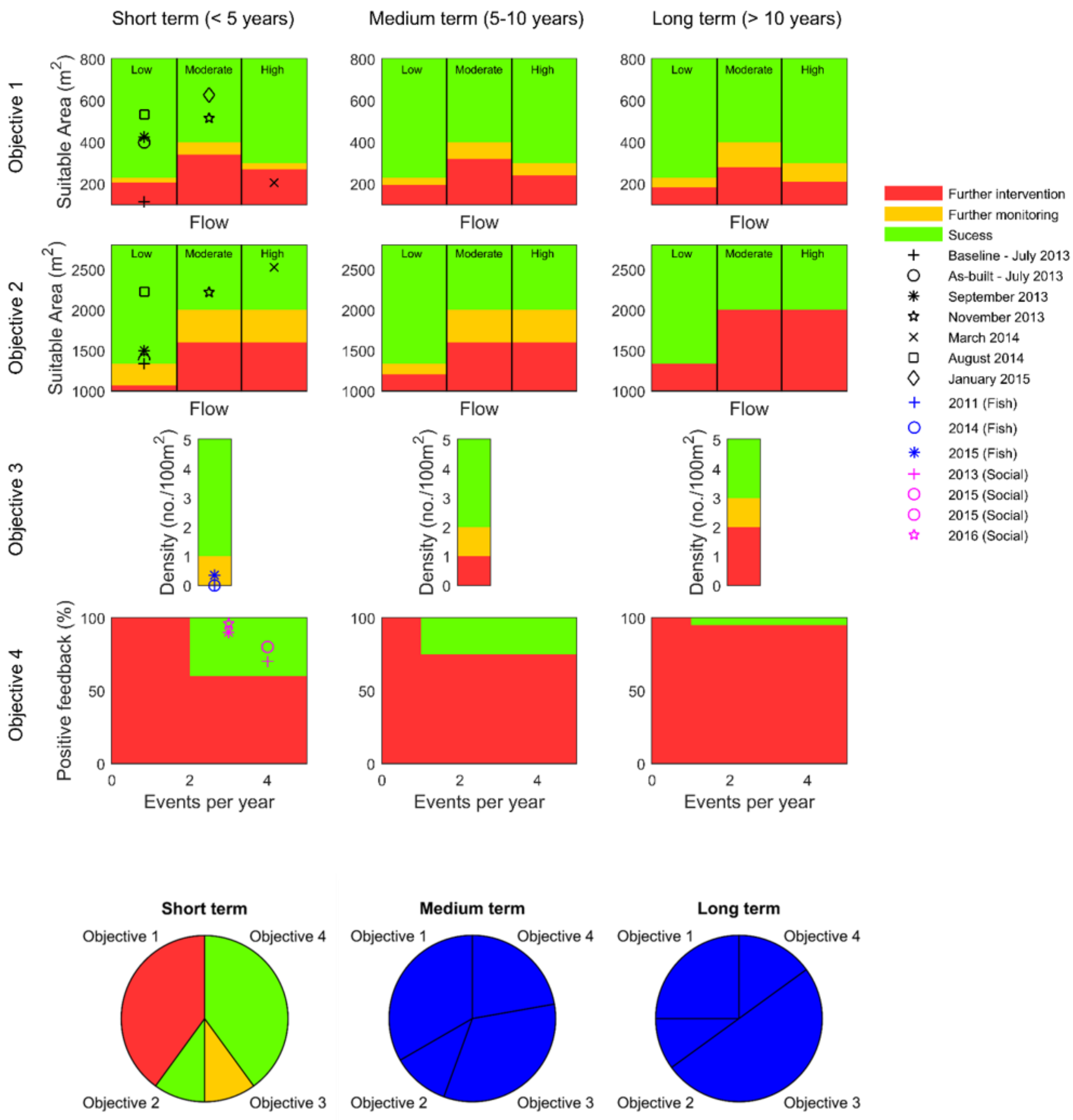


Figure E.1 Demonstration of the Performance Tracking Framework of a fictional scenario.

Appendix:

F. Ethical Review and Checklist

University of Portsmouth
Department of Geography

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Jennifer Cox
Department of Geography
29th November 2013

<u>Ethics review FAVOURABLE OPINION</u>

Protocol Title: Eco-hydraulic patch dynamics for river restoration schemes.

Dear Jennifer,

Thank you for submitting your Fast Track filter application form and your Major Review Protocol document for ethical review. This mode of review rather than the normal Faculty Committee review was adopted because there were no participants or significant ethics risks evident in your research. This route has been approved by the Faculty Committee as an interim measure until a University-wide ethics filter procedure is introduced.

Following review of your application I confirm that I can offer a favourable opinion on your proposals.

I undertook the review in October 2013, but I'm sending you this letter now as evidence of successful completion of the review.

Thank you for providing the additional information requested and I wish you good luck with the study.

Malcolm Bray

Dr Malcolm Bray
Department of Geography Ethics Co-ordinator
Science Faculty Ethics Committee Member

FORM UPR16

Research Ethics Review Checklist

University of Portsmouth

Please include this completed form as an appendix to your thesis (see the Postgraduate Research Student Handbook for more information

Postgraduate Research Student (PGRS) Information			**Student ID:**	449333
PGRS Name:	Jennifer Rose Cox			
Department:	Geography	**First Supervisor:**	Dr Philip Soar	
Start Date: (or progression date for Prof Doc students)		**1/10/2012**		
Study Mode and Route:	Part-time ☒ Full-time ☐	MPhil ☐ PhD ☒	MD ☐ Professional Doctorate ☐	

Title of Thesis:	Data-driven performance assessments for river restoration schemes (previously titled ecohydraulic patch dynamics for river restoration schemes on internal university documents.
Thesis Word Count: (excluding ancillary data)	76,996

If you are unsure about any of the following, please contact the local representative on your Faculty Ethics Committee for advice. Please note that it is your responsibility to follow the University's Ethics Policy and any relevant University, academic or professional guidelines in the conduct of your study

Although the Ethics Committee may have given your study a favourable opinion, the final responsibility for the ethical conduct of this work lies with the researcher(s).

UKRIO Finished Research Checklist:
(If you would like to know more about the checklist, please see your Faculty or Departmental Ethics Committee rep or see the online version of the full checklist at: http://www.ukrio.org/what-we-do/code-of-practice-for-research/)

a) Have all of your research and findings been reported accurately, honestly and within a reasonable time frame?	YES ☒ NO ☐
b) Have all contributions to knowledge been acknowledged?	YES ☒ NO ☐
c) Have you complied with all agreements relating to intellectual property, publication and authorship?	YES ☒ NO ☐
d) Has your research data been retained in a secure and accessible form and will it remain so for the required duration?	YES ☒ NO ☐
e) Does your research comply with all legal, ethical, and contractual requirements?	YES ☒ NO ☐

Candidate Statement:

I have considered the ethical dimensions of the above named research project, and have successfully obtained the necessary ethical approval(s)

Ethical review number(s) from Faculty Ethics Committee (or from NRES/SCREC):	No number was provided.

If you have *not* submitted your work for ethical review, and/or you have answered 'No' to one or more of questions a) to e), please explain below why this is so:

This work was submitted for Ethical Review to Dr Malcolm Bray who gave a favorable ethics review, please see copy attached.

UPR16 – August 2015

Signed *(PGRS)*:	**Date:** 26/08/2017